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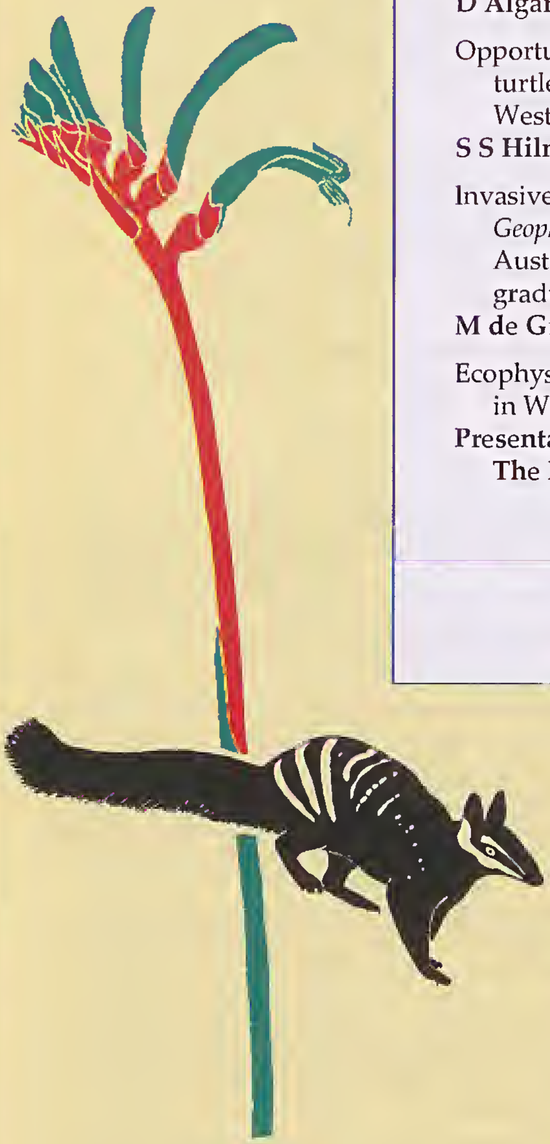

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Cover design: The four subjects symbolize the diversity of sciences embraced by the Royal Society of Western Australia. Mangles' kangaroo paw (*Anigozanthos manglesii*) and the numbat (*Myrmecobius fasciatus*) are the floral and faunal emblems of Western Australia, and stromatolites are of particular significance in Western Australian geology (artwork: Dr Jan Taylor). The Gogo Fish (*Mcnamaraspis kaprios*) is the fossil emblem of Western Australia (artwork: Danielle West after an original by John Long).

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Resistance of mammal assemblage structure to dryland salinity in a fragmented landscape

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Abstract

The impact of secondary environmental disturbance on native mammals is rarely considered. What little evidence exists, suggests that native mammal assemblages are resistant to secondary impacts. The Wheatbelt of Western Australia (WA) is a large fragmented region, resulting from historic land clearing for agriculture. Dryland salinity is a secondary environmental disturbance occurring in the Wheatbelt, as a result of land clearing, caused by rising groundwater tables and mobilisation of regolith salt to the soil surface. I used trapping and transect surveys to assess how the abundance of non-volant native, invasive and agricultural mammals related to salinity and other environmental attributes. I found support for the hypothesis that native mammals are resistant to the effects of compounding environmental change. I also found that abundance of introduced mammals were unrelated to salinity, except sheep *Ovis aries* (Linnaeus 1758, Bovidae), which exhibited reduced abundance in saline areas, relating to pasture availability. Instead, individual and assemblages of mammals exhibited associations with other environmental factors, which is consistent with other studies. Findings here support other studies which indicate native mammals that survive primary environmental change are resistant to further impacts. Resistance of native mammals to secondary environmental impacts may be a widespread phenomenon.

Keywords: Resistance, environmental change, dryland salinity, secondary salinisation, distribution, remnant vegetation, Western Australia, Wheatbelt

Introduction

The effect of discrete environmental impacts, such as deforestation and agriculture, on an assemblage of organisms has been widely studied. Specialist taxa, which are often native, tend to be negatively affected, whereas generalist and invasive taxa typically benefit (for example, Bennett 1993; Kaminski *et al.* 2007; Kitchener *et al.* 1980; Pearce & Venier 2005; Stephenson 1995; Suntsov *et al.* 2003; Torre *et al.* 2007; Umetsu & Pardini 2007). The effect of sequential environmental impacts on the same ecosystem has been considered less often, particularly for mammals. Of the few mammal studies that have considered sequential impacts, assemblages appear to be resistant to secondary environmental impacts (Larsen *et al.* 2007; Michalski & Peres 2007). For example, Larsen *et al.* (2007) found the impacts of timber harvesting and fire on small mammals were similar, but not additive; and Michalski & Peres (2007) found that assemblages of medium to large sized mammals which were already affected by fragmentation in tropical forests did not respond further to fire. Here, I examine the relationship of individual species and assemblages of mammals in a fragmented landscape, in the southwest of Western Australia (WA), to a secondary environmental impact, dryland salinity, and other environmental attributes.

Mammals form a charismatic part of the biota of the southwest corner of Western Australia, which is one of the world's 25 biodiversity hotspots (Keighery *et al.* 2004). The inland region of southwest WA (the Wheatbelt, 20 million hectares) has been extensively cleared for agriculture (grazing and cereal growing). Overall 74% of the natural land cover has been cleared (McKenzie *et al.* 2003; McKenzie *et al.* 2004) and assemblages of mammals that remain are composed of relic, agricultural and invasive species (Burbidge & McKenzie 1989). Seventeen native mammals are now considered extinct in the Wheatbelt region and at least 10 mammals have been either introduced for agriculture or are invasive (Burbidge & McKenzie 1989; Burbidge *et al.* 2004; Kitchener *et al.* 1980; McKenzie *et al.* 2002). A secondary consequence of land clearing in the Wheatbelt has been a rise in groundwater tables (due to a reduction in the number of deep rooted tree species that maintain low water tables via high rates of evapotransporation). Rising groundwater tables have mobilised large regolith salt deposits to the soil surface. There has been a twenty fold increase in salinity in the Wheatbelt (termed "dryland salinity") since European settlement (Halse *et*

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al. 2003). Currently more than one million hectares are affected by dryland salinity and this is expected to expand two to four fold by 2050 (George *et al.* 2006; Jardine *et al.* 2007).

Dryland salinity has numerous environmental impacts that are relevant to the ecology of mammals. Vegetation structure is impacted by salinity (Cramer & Hobbs 2002; Gibson *et al.* 2004; Lyons *et al.* 2004; Seddon *et al.* 2007), potentially affecting habitat for all mammals and food availability for herbivores. Salinity also influences the occurrence of vertebrate and invertebrate fauna (McKenzie *et al.* 2003; McKenzie *et al.* 2004), which may alter food availability for carnivorous, omnivorous and insectivorous mammals. Physiologically, saline water may also exert pressure on osmoregulation and the ability of mammals to persist and/or survive in saline areas (Kennear *et al.* 1968; Patrick *et al.* 2001; Purohit 1967, 1971). To date, few studies have examined the effects of saline habitats on mammal populations. Those studies that have documented associations between salinity and mammal populations have focused on small mammals and found the abundance and diversity of taxa to be unaffected by salinity (Burbidge *et al.* 2004; McKenzie *et al.* 2003; Smith *et al.* 1997). This study builds on previous research (Burbidge *et al.* 2004; McKenzie *et al.* 2003) by considering a range of sizes of non-volant terrestrial mammals and the relative influence of salinity on native and introduced species in a highly fragmented landscape. Dryland salinity represents a secondary environmental degradation for native mammals only.

In this study, I hypothesise: native and introduced mammal fauna will exhibit resistance to secondary disturbance of dryland salinity and assemblage structure is more related to other environmental variables that are known to affect their distribution and abundance. I inherently assume mammal communities measured here have adapted to fragmentation and, in saline sites, fragmentation and salinity. Thus the effect of secondary disturbance on native mammals equates to the difference in population structure between saline and non-saline area. I deem this assumption reasonable because regions surrounding all sites in this study have been fragmented for at least 40 years, and saline sites are known to have been saline for not fewer than five years (Burbidge & McKenzie 1989; Keighery *et al.* 2004 and references therein), which exceeds or approximately equates to the lifespan of many mammal species recorded here (Strahan 1988). Because mammal abundance may vary between seasons, I also examine this variability with respect to my hypothesis.

Methods

Study area

This study was undertaken in the Great Southern meteorological district of the WA Wheatbelt (Figure 1). The study area has a Mediterranean climate with hot dry summers and mild wet winters. Annual rainfall declines from approximately 600mm at the western boundary of my study sites to 350mm for the most easterly sites (Australian Bureau of Meteorology). Approximately 80–90% of the study area has been cleared for agriculture (Halse *et al.* 2004). Trapping and transect survey sites were separated by 5km or more, located in remnant native vegetation and were chosen to correspond with locations previously used by the WA Department of Environment and Conservation to assess the impacts of dryland salinity (Keighery *et al.* 2004).

Surveys

Surveys of mammals were undertaken using a combination of transect surveys and trapping across 12 sites (see Figure 1 for site names and coordinates). Surveys were conducted from September to December 2006 (spring) and February to May 2007 (late summer and autumn). Surveys followed ASM guidelines (Journal of Mammalogy 88: 809–823, 2007) and were ethically approved by the University of Western Australia Animal Ethics Committee (06/100/566).

Transect surveys were used to record mammal species which were unlikely to be captured by trapping, such as macropods, foxes and agricultural animals. Surveys were undertaken during the hour immediately after dawn and in the half hour before dusk, to correspond to the period of highest activity of macropods and other crepuscular species (Arnold *et al.* 1989; Hill *et al.* 1988; Priddel 1986). Surveys consisted of walking an established line transect through remnant vegetation or slowly driving a transect along a farm track/dirt road through or adjacent to the remnant. A minimum of one walking and one driving transect were performed at a site on each survey occasion. Transect surveys at each site were replicated 6–8 times (see Figure 1 for number of surveys at individual sites) and averaged to account for temporal and spatial variability in observations. Equal numbers of dawn and dusk surveys were conducted at each site. Consecutive surveys of a site were separated by a minimum of four days, to avoid survey bias in mammal behaviour and distribution. In total, 87 surveys were conducted across all sites, consisting of 233 individual transects.

On each transect, I measured the total distance [with a calibrated pedometer (VIA™ Step, New Balance®, Fenton, Missouri) or vehicle odometer (± 10 meters)], bearing, observational area within each habitat type and entire transect [transect length within habitat type x distance over which sight-ability of animals was reliable for each habitat type (50 meters in vegetation and forestry and 200 meters on roads or across agricultural land) summed for the total transect], mammal species, distance and bearing to each observed mammal. To minimise counting the same animal twice, transects were separated by greater than 200 metres. In rare cases where an animal was suspected of being counted twice, the result was discarded from one of the surveys. Sheep, *Ovis aries* (Linnaeus 1758, Bovidae) that were in paddocks surrounding the remnant vegetation were also counted during surveys. Numbers of sheep were approximated when individuals could not be reliable counted (typically when flocks exceeded 50). The abundance of mammals at each site was calculated as the number observed per hectare (number observed/observational area across all transects).

Trapping grids were laid in areas of remnant vegetation at each site. I used box traps (90x90x350mm, Elliott Scientific Equipment, Upwey, Victoria) and wire cage traps (220x220x550mm, Sheffield Wire Products, Welshpool WA). Both trap-types were baited with peanut

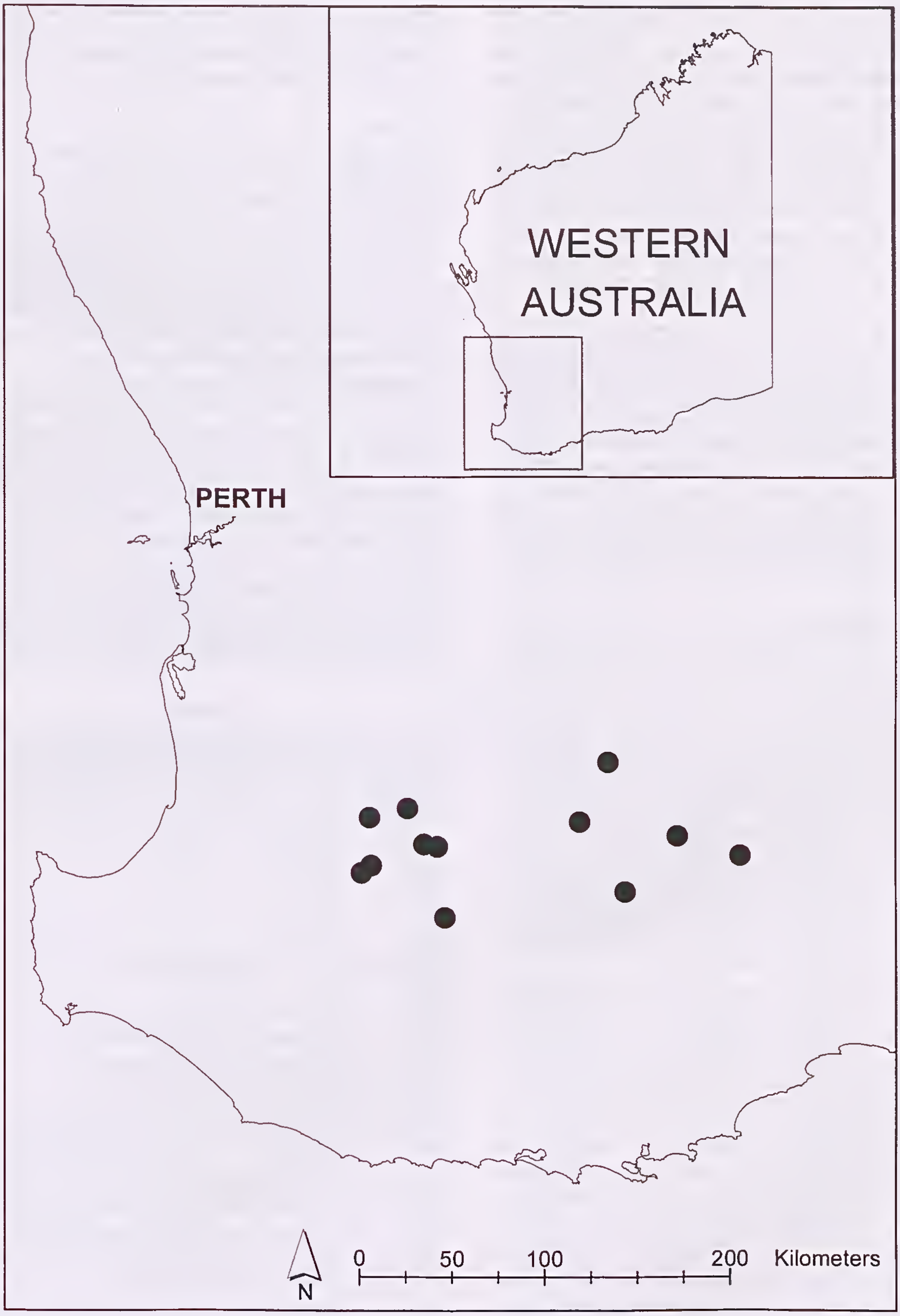

Figure 1. Sites surveyed for mammal fauna in the Wheatbelt of southwest WA. Site names, their coordinates and survey effort per site (number of transect and trapping surveys respectively): DA02, lat -33.35628 long 116.62067 (7, 2); DA03, lat -33.31329 long 116.80671 (7, 2); DA05, lat -33.48809 long 116.88669 (7, 2); DA07, lat -33.49884 long 116.95164 (8, 2); DA10, lat -33.59043 long 116.63058 (7, 2); DA12, lat -33.62621 long 116.58273 (7, 2); DA13, lat -33.84380 long 116.98703 (8, 2); DU02, lat -33.08636 long 117.77947 (6, 2); DU06, lat -33.37755 long 117.64134 (8, 2); DU08, lat -33.71999 long 117.86499 (8, 2); DU09, lat -33.44393 long 118.11712 (8, 2); DU13, lat -33.54058, long 118.42258 (6, 2).

butter (Sanitarium®, New South Wales) and rolled oats (Asia/Pacific Wholesalers Pty. Ltd., Parramatta, New South Wales). Box traps were set on a 7x10 grid (n = 70) with 8 metres between each row of traps. Cage traps were placed next to every second box trap along four of the trap lines (n = 20). Nesting material was used to insulate box traps; each cage trap was insulated by covering with a hessian bag. Traps were set just before dusk, then cleared and closed shortly after dawn. Each site was trapped twice (once in spring and once in autumn) for three consecutive nights, giving a total of 6480 trap nights. Trapped animals were identified to species, marked using blue food colouring (Queen Fine Foods Pty. Ltd., Alderley, Queensland) and released.

Trapping was used to estimate trap success (the number of unique individuals trapped per 1000 trap nights) of any mammal within the trapping grid. Trap success, or the related abundance estimate "minimum number alive", are conservative measures of abundance for small mammals, and common measures of abundance used for species such as the introduced domestic mouse *Mus domesticus* (Schwarz & Schwarz 1943, Muridae: syn. *Mus musculus* Linneaus 1758; McKenzie *et al.* 2003; Ruscoe *et al.* 2001). At all sites, the maximum diversity of trapped mammals was achieved within the first two nights of trapping.

Environmental data

To examine how mammals were associated to salinity and other environmental data, soil (n = 8), vegetation (n = 5) and geomorphic (n = 5) variables, for each of the 12 sites, were obtained from the WA Department of Environment and Conservation (Keighery *et al.* 2004 and references therein: see Table 1 for variables and definitions). Briefly: soil variables consisted of standard texture and chemical values collected at a depth of 2–10cm (after removal of surface litter) from 20–30 points per site, then bulked for analysis; vegetation variables were subjective assessments of habitat, divided into categories, using methods modified from Newsome & Catling (1979); geomorphic variables included subjective categories of salinity impacts and risk, and variables relating to position in the landscape.

Vegetation and geomorphic variables, despite being subjective, provided good estimates of habitat (within remnants) and position in the landscape. Categories of salinity impacts (adjSAL and Saltype, Table 1) at sites followed those described by Keighery *et al.* (2004) and references therein, and further developed by Carver *et al.* (2009) to encompass the broader area (within and surrounding remnants) in which surveys were conducted. Briefly, adjSAL: (1) non-saline, due to absence of stags and samphire, high position in the landscape, low water table, high soil permeability, and/or the low salt store in the regolith (n = 4); (2) low salinity impact or risk, due to stags and samphire being uncommon, small variation in local relief and geology where rising water tables may not affect all the land area, or where rising watertables are not currently saline, and the salt store in the regolith is low (n = 2); (3) intermediate salinity impacted, due to common occurrence of stags and samphire, salinity occurring in limited areas, or from shallow saline groundwater with a rising trend (n = 1); and (4) saline land, due to widespread occurrence of stags and samphire and obvious salt effects (such as large patches of bare earth) in the area (n = 5). For Saltype (Table 1) sites were specifically categorised as: not saline (n = 6) or salinity affected (n = 6).

Analyses

For mammals that occurred at three or more sites, I initially examined if my estimates of mammal abundance differed between spring and autumn for both transect and trapping surveys (number of mammals surveyed per hectare and trap success per 1000 trap nights referred to as abundance here forth). Pairwise student t-tests were used to examine differences between spring and autumn trapping sessions. Abundance of mammals did not differ between seasons (see results). Data for each species were pooled and used for all subsequent analyses.

Analyses of the relationship of assemblages and species of mammals to salinity, and other environmental variables, were based on the assumption that variation in the spatial abundance of mammals and their assemblages reflect underlying correlations with environmental factors (Austin 1991). Dryland salinity is known to influence plant species occurrence, abundance and species composition (Cramer & Hobbs 2002; George *et al.* 1999; Lymbery *et al.* 2003; Seddon *et al.* 2007). This interrelationship may confound analyses between salinity, vegetation and abundance of mammals. Accordingly, relationships among environmental variables were assessed by Spearman correlation.

Table 1

Environmental attribute codes. Data and codes derived from Keighery *et al.* (2004) and references therein, with permission from authors. adjSAL and Saltyp further developed by Carver *et al.* (2009).

Code	Attribute
Clay	% clay
Silt	% silt
Sand	% sand
Drainage	Soil drainage in six classes (poor – rapid)
pH	
OrgC	Organic carbon %
P	Available (HCO_3) phosphorus (ppm)
K	Available (HCO_3) potassium (ppm)
TreeCanp	Tree canopy in three categories
ShrubCov	Shrub cover in three increasing categories
GrndHerb	Ground herbs in three categories (sparse – dense)
GrndStuf	Ground cover by leaf litter, rocks and fallen wood in three increasing categories
HCS	Habitat complexity score in ten categories
Landform	Twelve position categories (valley floor and deposits rising to plateau)
Slope	Slope in five categories (flat – steep)
EC	Electrical conductivity (mSm^{-1})
adjSAL	Salinity in four classes (non-saline, low salinity, intermediate salinity, saline)
Saltyp	Salinity type in two classes (not saline, salinity affected)

Measures of vegetation in this study were of general structure and not taxa specific (see Table 1). For the singular case where a vegetation variable was related to a salinity variable (GrndStuf and Saltyp, see Table 2), subsequent comparisons of mammal data with ground cover were made using the residuals.

The distribution and abundance of *O. aries*, in this study, is influenced by intensive management. Nevertheless, the distribution and abundance of this species is still constrained by environmental and physiological limitations (*e.g.*, food production and osmoregulation in saline environments). Therefore, *O. aries* were included in analysis of geomorphic and soil variables, but vegetation variables were not analysed as sheep did not occur within remnant vegetation.

The similarity of mammal assemblages between sites was examined by Non-Metric Multidimensional Scaling (NMDS) in PRIMER v6 (PRIMER-E Ltd: Plymouth, U.K.). NMDS were carried out on Bray-Curtis similarity matrices from square root transformed and standardised abundance data. A Spearman rank correlation (ρ: the BIOENV procedure in PRIMER v6) was used to assess which combination of environmental variables best explained the multivariate variation in assemblage structure.

Relationships between abundance of individual mammal species, for species that occurred in three or more sites, and environmental variables were examined using linear regression. Additionally, a power analysis was used to determine the probability of detecting a significant relationship between salinity and abundance of mammals. For parametric univariate analyses in this study, data were examined for normality (Shapiro-Wilk's test of normality) and equal variances (Levene's test) prior to analysis. Data which were not normal or had unequal variances were normalised by log transformation. Analyses were undertaken using SPSS 15.0 (SPSS Inc. Chicago, U.S.). To avoid committing Type-II statistical error, analyses were not adjusted for multiple comparisons (Nakagawa 2004).

Results

Environmental variables

Salinity was greater at sites which were flat, low in the landscape and with poor drainage (Table 2). General measures of vegetation were unrelated to site salinity, except ground cover, which was negatively related to salinity type (Table 2)

The occurrence and abundance of mammals

Eight mammals were observed during transect surveys [native species: Western Grey kangaroo *Macropus fuliginosus* (Desmarest 1817, Macropodidae); Brush wallaby *Macropus irma* (Jourdan 1837, Macropodidae), and introduced species: rabbit *Oryctolagos cuniculus* (Linnaeus 1758, Leporidae); sheep

Table 2

Significant Spearman correlations between environmental variables. See Table 1 for definitions.

Variable	Correlation	ρ	P
Clay	Sand	-0.713	0.009
	pH	0.611	0.035
Silt	Sand	-0.697	0.012
	P	0.763	0.004
	K	0.767	0.004
	EC	0.746	0.005
	Drainage	-0.583	0.047
	Landform	-0.605	0.037
Sand	Clay	-0.713	0.009
	Silt	-0.697	0.012
	pH	-0.730	0.007
	K	-0.622	0.031
	ShrubCov	-0.637	0.026
Drainage	Silt	-0.583	0.047
	K	-0.790	0.002
	EC	-0.768	0.004
	Landform	0.725	0.008
pH	Clay	0.611	0.035
	Sand	-0.730	0.007
	Saltyp	0.606	0.037
	GrndStuf	-0.662	0.019
OrgC	P	0.597	0.041
P	OrgC	0.597	0.041
	EC	0.754	0.005
K	Silt	0.767	0.004
	Sand	-0.622	0.031
	EC	0.676	0.016
	Drainage	-0.790	0.002
ShrubCov	Sand	-0.637	0.026
GrndHerb	HCS	0.650	0.022
GrndStuf	pH	-0.662	0.019
	Saltyp	-0.617	0.033
	HCS	0.607	0.036
HCS	GrndHerb	0.650	0.022
	GrndStuf	0.607	0.036
Landform	Silt	-0.605	0.037
	EC	-0.630	0.028
	Drainage	0.725	0.008
Slope	adjSAL	-0.702	0.011
EC	Silt	0.746	0.005
	P	0.754	0.005
	K	0.676	0.016
	Saltyp	0.580	0.048
	Drainage	-0.768	0.004
	Landform	-0.630	0.028
adjSAL	Saltyp	0.920	<0.001
	Slope	-0.702	0.011
Saltyp	pH	0.606	0.037
	EC	0.580	0.048
	adjSAL	0.920	<0.001
	GrndStuf	-0.617	0.033

Table 3

For mammals that occurred at three or more sites: the number of sites each mammal species was detected and average abundance (number per hectare) and trap success (number/1000 trap nights) (mean ± SE) in spring and autumn. Student t-tests to detect differences in abundance and trap success between seasons.

	Spring Occurrence	Abundance/trap success Mean	SE	Autumn Occurrence	Abundance/trap success Mean	SE	$t_{1,22}$	*P*
M. fuliginosus	12	0.199	0.04	12	0.239	0.066	0.483	0.634
O. cuniculus	9	0.015	0.005	7	0.009	0.005	0.862	0.398
O. aries	11	1.516	0.338	12	1.164	0.182	0.959	0.348
V. vulpes	2	0.001	0.001	6	0.004	0.001	1.323	0.199
M. domesticus	9	55.952	16.779	7	22.619	7.891	1.258	0.222
T. vulpecula	2	20.833	17.556	3	25.000	17.604	0.175	0.863

O. aries; goat *Capra hircus* (Linnaeus 1758, Bovidae); horse *Equus caballus* (Linnaeus 1758, Equidae); pig *Sus scrofa* (Linnaeus 1758, Suidae); and fox *Vulpes vulpes* (Linnaeus 1758, Canidae)] and four species were trapped [native species: Brushtail possum *Trichosurus vulpecula* (Kerr 1792, Phalangeridae); Mitchell's hopping mouse *Notomys mitchelli* (Ogilby 1838, Muridae); Red-tailed phascogale *Phascogale calura* (Gould 1844, Dasyuridae), and the introduced house mouse, *Mus domesticus*]. *Ovis aries*, *Capra hircus* and *Equus caballus* were agricultural animals and the remaining species were either wild native or feral. No mammal was recorded by both transect and trapping surveys. Relatively few native mammals were trapped or observed. *M. fuliginosus* was the most widespread native species recorded (Table 3). *Trichosurus vulpecula* was recorded at the highest abundance for native mammals, but was only trapped at three sites (Table 3). *Macropus irma* was recorded in autumn at one site only. *Notomys mitchelli* and *P. calura* were both abundant (> 0.5 per hectare), but occurred at only one and two sites respectively. Of the invasive and agricultural mammals, *O. aries* were the most widespread and *Mus musculus* the most abundant (Table 3). The remaining introduced species (*C. hircus*, *E. caballus*, *S. scrofa*) were uncommon (occurring at one site each), except for *O. cuniculus* and *V. vulpes*, which were detected infrequently at ten and six of the twelve sites, respectively (Table 3). For both trapping and transect surveys, the average abundance of each species did not different between spring and autumn (Table 3).

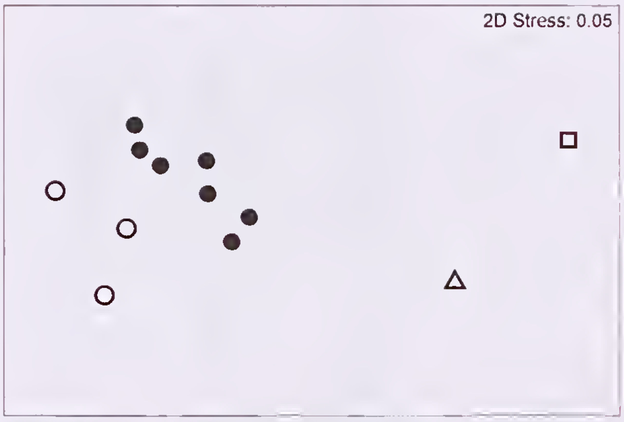

Figure 2. NMDS of mammal assemblages (n = 12) based on trapping and transect data combined. Sites that are circles are similar in assemblage composition, with open circles representing sites in which *T. vulpecula* occurred. The open square and triangle sites differed from other sites, due to the absence of *M. domesticus* and presence of *P. calura*. The open square represents a site that also contained *N. mitchelli*.

Mammals, salinity and other environmental variables

Assemblage structure of mammals was similar for ten of the twelve sites (clustered together on the NMDS plot, Figure 2), predominantly composing of *M. fuliginosus*, *O. cuniculus*, *O. aries* and *M. domesticus*. Of these ten sites, three appeared distinct, likely due to the additional occurrence of *T. vulpecula* (Figure 2). Assemblage structure of mammals for the remaining two sites was distinct, due to the absence of *M. domesticus* and presence *P. calura* (Figure 2). These two sites were different from each other, due to the presence of *N. mitchelli* at one site (Figure 2). The assemblage structure of mammals was best explained by three environmental variables (Spearman correlation from BIOENV procedure, ρ = 0.460): phosphorus, slope and ground herb cover.

For mammals which occurred at three or more sites, abundance of *O. aries* only were related, negatively, to increasing salinity category (Table 4, Figure 3c). The probability of detecting a significant effect of salinity on abundance of most mammal species was small, with the

Table 4

Significant linear regressions between the abundance (number per hectare) and trap success (number/1000 trap nights) of mammals at sites (n = 12) and environmental variables. See Table 1 for definitions of environmental attribute codes.

	Predictor	r^2	Regression Coefficient	SE	$F_{1,10}$	*P*
WGK*	GrndHerb	0.486	0.119	0.039	9.465	0.012
Rabbit	Clay	0.400	0.029	0.011	6.663	0.027
	Sand	0.338	-0.001	<0.001	5.108	0.047
Sheep	EC	0.339	-0.409	0.181	5.123	0.047
	adjSAL	0.720	-0.418	0.083	25.665	<0.001
	Saltyp	0.696	-1.086	0.227	22.898	<0.001
	Slope	0.453	0.702	0.244	8.270	0.017
Mouse	GrndStuf	0.452	35.204	12.267	8.236	0.017
	K	0.331	61.362	27.557	4.958	0.050
	Drainage	0.377	-12.663	5.136	6.051	0.034
Possum	Silt	0.470	172.377	57.899	8.864	0.014
	Sand	0.807	-8.320	1.286	41.865	<0.001
	P	0.935	7.702	0.642	143.904	<0.001

* WGK = Western Grey kangaroo, *M. fuliginosus*

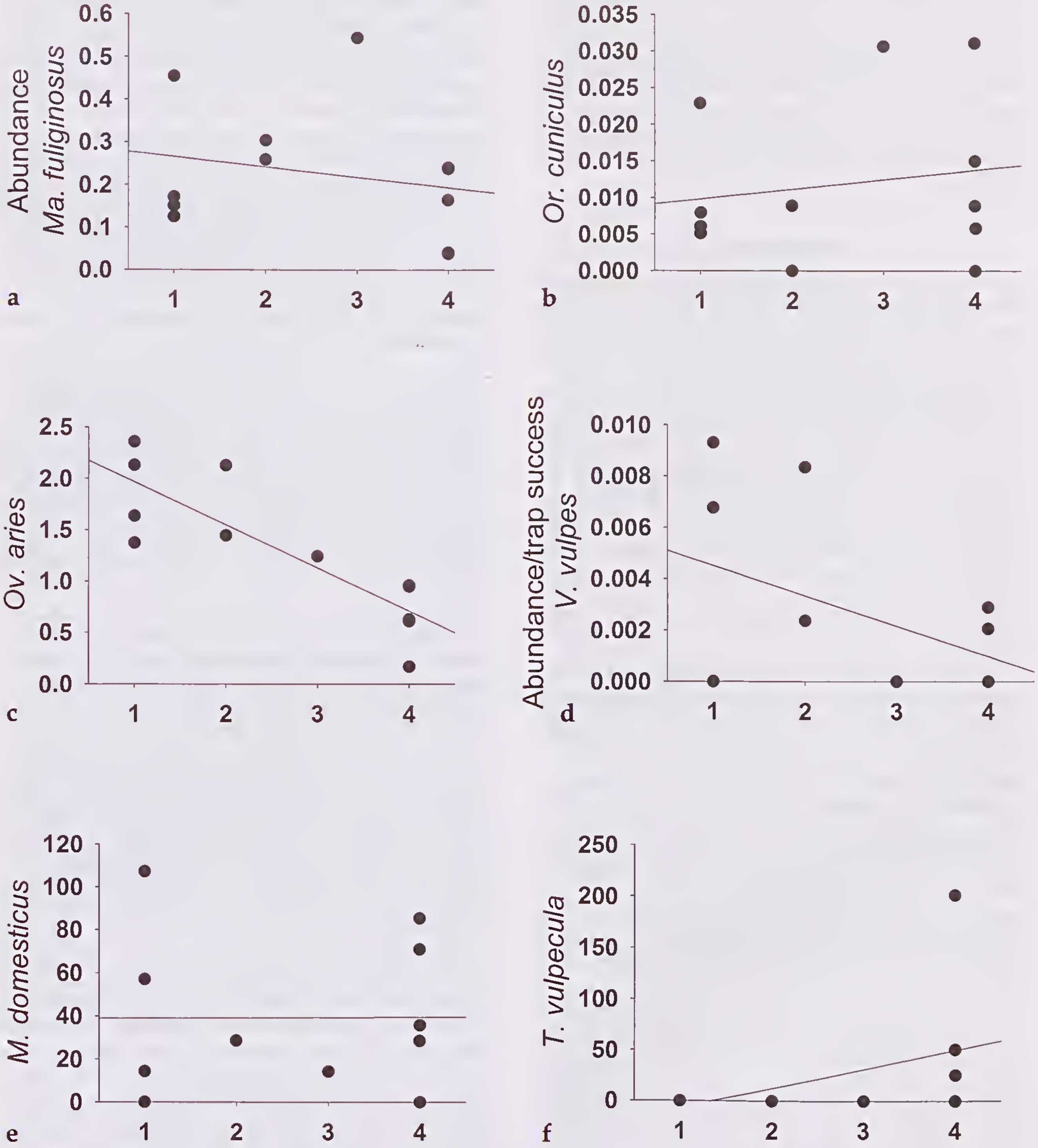

Figure 3. Relationships between the abundance (number per hectare) and trap success (number/1000 trap nights) of mammals with adjSAL category (Table 1). Linear regressions and power analysis for probability of detecting a significant salinity effect: (a) $r^2 = 0.049$, $F_{1,10} = 0.517$ $P = 0.488$, power = 0.106; (b) $r^2 = 0.028$, $F_{1,10} = 0.286$ $P = 0.640$, power = 0.079; (c) $r^2 = 0.720$, $F_{1,10} = 25.665$ $P < 0.001$ (regression coefficients (± SE), predictor -0.418(0.083) and constant 2.385(0.240)) , power = 0.999; (d) $r^2 = 0.214$, $F_{1,10} = 2.720$ $P = 0.130$, power = 0.428; (e) $r^2 = 0.001$, $F_{1,10} = 0.001$ $P = 0.983$, power = 0.046; and (f) $r^2 = 0.197$, $F_{1,10} = 2.454$ $P = 0.148$, power = 0.346.

possible exception of *V. vulpes* (Figure 3), suggesting there was little discernable relationship between salinity and abundance of most mammals (except *O. aries*) in this study. Abundance of *M. fuliginosus* and *M. domesticus* were related to vegetation variables (Table 4). Abundance of *V. vulpes* were not significantly related to any environmental variable (Table 4, Figure 3d), but there was a positive trend with ground cover ($r^2 = 0.005$, $F_{1,10} = 4.947$ $P = 0.0503$). Abundance of *O. cuniculus*, *M. domesticus* and *T. vulpecula* were related to soil variables (Table 4).

Discussion

Few studies consider the effect of secondary environmental impacts on mammals. Here, I evaluated how abundance of mammals was related to dryland salinity (a secondary environmental impact) and other environmental variables in a fragmented landscape. Overall, the abundance of native mammals, for which dryland salinity represents a secondary environmental impact, was not associated with salinity, suggesting that these taxa are resistant to the secondary environmental degradation, brought about by dryland salinity. In contrast, salinity was explanatory of *O. aries* abundance. It is acknowledged that *O. aries* are extensively managed and, as such, my estimates of their abundance reflect management. This relationship is unsurprising, because saline land supports lesser amounts of pasture and consequently agricultural production is lower (Lloyd 2001).

Widespread clearing and habitat fragmentation are characteristic features of the Wheatbelt (Keighery *et al.* 2004). Generally the relic native fauna that remain are indicative of this disturbance (Burbidge & McKenzie 1989; Kitchener *et al.* 1980; McKenzie *et al.* 2007) (but see Abbott 2006). Of the native mammals that I detected, *M. fuliginosus* was widespread, *T. vulpecula* occurred at one quarter of the sites and the remaining native mammals were seldom encountered. All other mammals were either agricultural or invasive, with the most frequent of these being *O. aries*, *O. cuniculus*, *V. vulpes* and *M. domesticus*. Native mammals which remain in the Wheatbelt are species that have adapted to, exploited or tolerated the primary environmental disturbance of habitat clearing and fragmentation and according to my results, do not appear to be affected by dryland salinity.

My results are supported by Smith *et al.* (1997) who found that the abundance of *M. domesticus*, *Sminthopsis crassicaudata* (Gould 1844, Dasyuridae) and *Sminthopsis dolichura* (Kitchener, Stoddart & Henry 1984, Dasyuridae) did not decline on saline land. Burbidge *et al.* (2004), also found richness of mammal fauna was unrelated to salinity. However, trapping occurred on naturally saline land for both these studies, rather than anthropogenic saline land. McKenzie *et al.* (2003) focussed on natural and anthropogenic saline land and also found that the richness of mammals and occurrence of *M. domesticus*, *Tarsipes rostratus* (Gervail & Verraux 1842, Tarsipedidae) and *Sminthopsis dolichura* were unrelated to salinity. However, McKenzie *et al.* (2003) did find some native mammals were related to salinity. For example: *Cercartetus concinnus* (Gould 1845, Burramyidae) was widespread, but occurred rarely on saltflats; *Sminthopsis gilberti* (Kitchener, Stoddart & Henry 1984, Dasyuridae) and *Sminthopsis griseoventer* (Kitchener, Stoddart & Henry 1984, Dasyuridae) were absent from saltflats; and *Sminthopsis crassicaudata* was only recorded from saltflat environments. It is possible that in these cases, relationships with salinity may have been related to species specific food or habitat requirements, or interspecific interactions.

As predicted, assemblages of mammals were generally associated with other environmental factors, rather than salinity. For example, *M. fuliginosus* are positively related to the cover of ground herbs; and *M. domesticus* densities increased with ground cover, leaf litter, rock and woody debris. These habitat variables are indicative of a combination of food availability and shelter (Burbidge *et al.* 2004). However, in some cases it is difficult to determine the significance of associations with some environmental variables (Burbidge *et al.* 2004). For example, abundance of *M. domesticus* and *T. vulpecula* increased positively with potassium and phosphorus respectively. It is possible these associations may be indicative of resource and microhabitat differences between sites, but these factors were not identified in the variables examined here. With the obvious exception of *O. aries*, results here generally support the hypothesis; that mammal fauna exhibit resistance to the disturbance wrought by dryland salinity, and are related to other determinants of distribution and abundance, such as vegetation.

This study focussed on relationships between environmental variables and mammal abundance. However, interspecific interactions may also influence mammal occurrence and abundance. For example, *P. calura* is known to predate on *Mus domesticus* (Strahan 1988) and, accordingly, *M. domesticus* were not detected at the two sites occupied by *P. calura*. *Vulpes vulpes* also predates on other mammals, such as *O. aries*, *O. cuniculus* and *T. vulpecula* (Banks 2000; Greentree *et al.* 2000; McKenzie *et al.* 2007). However, I did not detect a negative abundance dependent relationship between *V. vulpes* and any of these species (linear regressions: $r^2 = 0.147$, $F_{1,10} = 1.719$, $P = 0.219$; $r^2 = 0.010$, $F_{1,10} = 0.096$, $P = 0.763$; and $r^2 = 0.0.028$, $F_{1,10} = 0.288$, $P = 0.603$, respectively).

There are some cases where native species may not have been detected in this study. For example, my trapping did not specifically target insectivorous or carnivorous mammals. This potentially excludes observing widespread native insectivores, like Dunnarts *Sminthopsis* sp. (Dasyuridae) and Quenda *Isoodon obesulus* (Shaw 1797, Peramelidae), and native carnivores, like the Western Quoll *Dasyurus geoffroyi* (Gould 1841, Dasyuridae) (McKenzie *et al.* 2003; Menkhorst & Knight 2001; Smith *et al.* 1997). However, *Sminthopsis* sp., *I. obesulus* and *D. geoffroyi* are still occasionally trapped using the methodology employed here (Chambers & Dickman 2002; Morris *et al.* 2003; Wilson *et al.* 1986), and it is probable these species were rare or absent at my sites. The WA Department of Environment and Conservation similarly did not detect any of these mammals at the same sites (Keighery *et al.* 2004).

Additionally, increased understorey vegetation may have obscured spotting mammals and thus have led to underestimates of abundance. But, every effort was made to be meticulous with observations during transect surveys and I did not detect a relationship between mammal abundance and observational area per survey for any species (Carver unpublished data). Furthermore, relationships between climatic variables and abundance of mammal species are not presented in this study. In preliminary investigations, I examined relationships between mammal abundance and climatic variables and found a weak relationship ($r^2 = 0.378$) between abundance of *M. fuliginosus* and average winter temperature between sites only, which was difficult to interpret and possibly spurious (climatic variables derived from each site using ANUCLIM, see Keighery *et al.* 2004 and McMahon *et al.* 1995). Accordingly, analyses of climate and abundance of mammal species between sites was excluded from this study.

Conclusions

The effect of multiple environmental impacts on mammals has received relatively little attention. This study hypothesised and found that non-volant native mammals were resistant to the effects of a compounding environmental change, dryland salinity. Abundance of introduced mammals was also unrelated to salinity, except *O. aries*, which exhibited reduced abundance in saline areas. In general, individual species and assemblages of mammals exhibited associations with other environmental factors. This study supports other studies, which have considered multiple impacts, concluding assemblages of native mammals appear to be resistant to secondary environmental impacts. Resistance of mammals, which survive primary environmental changes, to further environmental change may be a widespread phenomenon.

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Longevity in Australian snapper *Pagrus auratus* (Sparidae)

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Abstract

In fisheries biology, longevity is one of the estimators of natural mortality, an essential parameter for age-based stock assessments. Increased fishing pressure progressively truncates older fish from the population, highlighting the importance of documenting observations of exceptionally old individuals. Snapper (*Pagrus auratus*) constitute important fisheries around the southern half of mainland Australia. A literature search and survey of government and semi-government agencies responsible for managing snapper fisheries in 5 Australian states reveals the species is capable of living to at least 40 years throughout much of its range. The oldest recorded age, from near Bunbury, Western Australia, was recently estimated to be 40 years and 10 months.

Keywords: longevity, snapper, *Pagrus auratus*, age, mortality, otolith

Introduction

Potential longevity is inversely related to natural mortality. In fisheries biology, maximum age can be used to estimate the natural mortality rate (Hoenig 1983; Hewitt & Hoenig 2005), an essential parameter in many mathematical models of fish stock dynamics (Vetter 1988). Unfortunately the use of maximum age can become problematic in many fisheries because fishing pressure has already removed older fish from the population (Berkeley *et al.* 2004) before the age of exceptionally old individuals can be recorded.

In Australia, snapper (*Pagrus auratus*) is an iconic species that has a continuous distribution around the southern half of the mainland. It supports important commercial and recreational fisheries throughout its range (Kailola *et al.* 1993). Otolith based age estimates of Australian snapper show that it can live to over 30 years of age (McGlennon *et al.* 2000; Coutin *et al.* 2003; Wakefield 2006; Jackson 2007; Wise *et al.* 2007; Lenanton *et al.* 2009).

Documenting the age of exceptionally old snapper before they are removed by fishing preserves the capacity to estimate natural mortality from maximum age, perpetually enhancing the accuracy of age-based fishery assessments and helping to avoid a potential "shifting baseline syndrome" (Pauly 1995) in longevity reference points. This short note documents observations of exceptionally old snapper from wild Australian fisheries for use in mathematical stock assessment models.

Methods

The accepted method for ageing Australian snapper is to interpret the alternating opaque and translucent zones delineated annually in sagittal otoliths, combined with a nominal birth-date during peak spawning gauged from a time series of gonad developmental stages and a gonadosomatic index (McGlennon *et al.* 2000; Ferrell & Sumpton 1998; Coutin *et al.* 2003; Wakefield 2006; Jackson 2007; Lenanton *et al.* 2009). This method was used for all ages reported in this study.

The investigation to find Australia's oldest recorded snapper involved consulting published literature that reported data on snapper ages. Furthermore, government and semi-government agencies in each Australian state responsible for managing snapper fisheries and monitoring the age structure of catches were contacted in May 2009 and asked to provide details of the oldest snapper recorded in their respective states from unpublished reports or databases. These agencies were Queensland Primary Industries and Fisheries (QPIF), the New South Wales Department of Primary Industries (NSWDPI), Victoria's Department of Primary Industries (VDPI) in association with the Victorian based Fish Ageing Services Pty Ltd (FAS), the South Australian Research and Development Institute (SARDI), and the Department of Fisheries Western Australia (DFWA).

The instantaneous rate of natural mortality (*M*) was estimated from the age of the oldest recorded Australian snapper using:

1. Hoenig's (1983) regression equation for fish;
2. Hewitt & Hoenig's (2005) rule-of-thumb approach using their estimated appropriate value for the proportion of animals in the stock that survive to the maximum age: $P = 0.015$; and
3. Hewitt *et al.*'s (2007) indirect method No. 3, using the von Bertalanffy growth parameter estimate of $K = 0.14$ from Lenanton *et al.* (2009).

Results

Very old snapper with estimated ages of approximately 40 years were recorded from Victoria and Western Australia, and only slightly younger ages of

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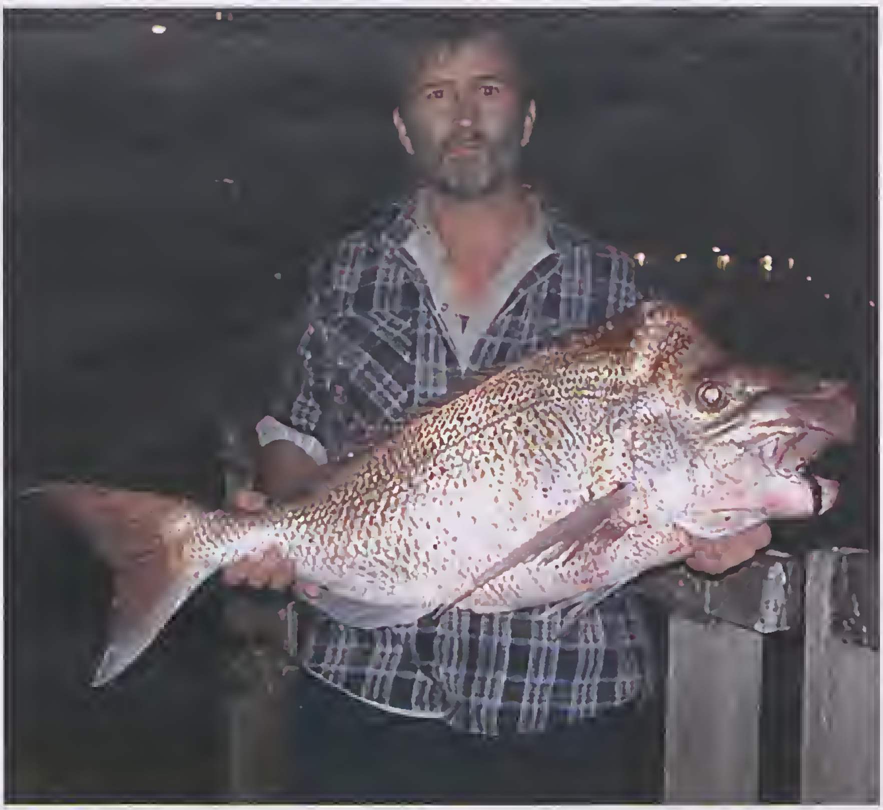

Figure 1. Australia's oldest recorded snapper (40 years and 10 months) captured approximately 9 nautical miles west of Bunbury, Western Australia.

about 37 years were reported from South Australia and New South Wales (Table 1). Several annuli in the otolith of the oldest snapper from New South Wales were indistinct, leading to some imprecision among repeated counts.

Australia's oldest recorded individual snapper (Figure 1) was aged at 40 years and 10 months, generating a natural mortality estimate of $M = 0.10$ using both the methods of Hoenig (1983) and Hewitt & Hoenig (2005), and an estimate of $M = 0.05$ using Hewitt *et al.*'s (2007) method. The fish was captured on 1 September 2007 and therefore a member of the 1966 year class. The otolith had 41 delineated opaque zones and a translucent margin less than 50% of the width of the previous translucent zone (Figure 2), indicating that the last formed opaque zone had delineated recently. The total length and length to caudal fork were 935 and 826 mm, respectively.

Table 1

Summary of the oldest recorded snapper from five Australian states. Source acronyms given in Methods.

Age (yrs)	Sex	Date of capture	State and location of capture	Source
23	M	15 Oct 2007	QLD, Brisbane	QPIF
ca 37	?	19 Oct 1994	NSW, northern coastal	NSWDPI
40.4	M	9 May 2001	VIC, Port Phillip Bay	VDPI, FAS
36.9	M	16 Dec 2000	SA, north Spencer Gulf	SARDI
40.8	M	1 Sep 2007	WA, Bunbury	DFWA

Discussion

Snapper of approximately 40 years of age from two widely separated locations in Victoria and Western Australia, and only slightly younger fish from South Australia and New South Wales, suggest the species can live to at least 40 years of age throughout much of its range. New South Wales had the largest annual snapper catch of all states from 1964–65 to 1982–83 (Kailola *et al.* 1993), a level of fishing which may have removed very old individuals, suggesting that longevity potential in that state may be higher than the approximately 37 years observed.

The low estimates of natural mortality for snapper throughout much of their range indicate this species is susceptible to over-fishing (Adams 1980). If stocks are fished at high levels the number of exceptionally old individuals is likely to decline in many locations and may not be encountered again. The longevity observations and natural mortality estimates in this study therefore form reference points that set a baseline for future age-based snapper fishery assessments.

Footnote: Recently Gomon *et al.* (2008) referred to snapper as *Chrysophrys auratus*, citing genetic research by Orrell & Carpenter (2004).

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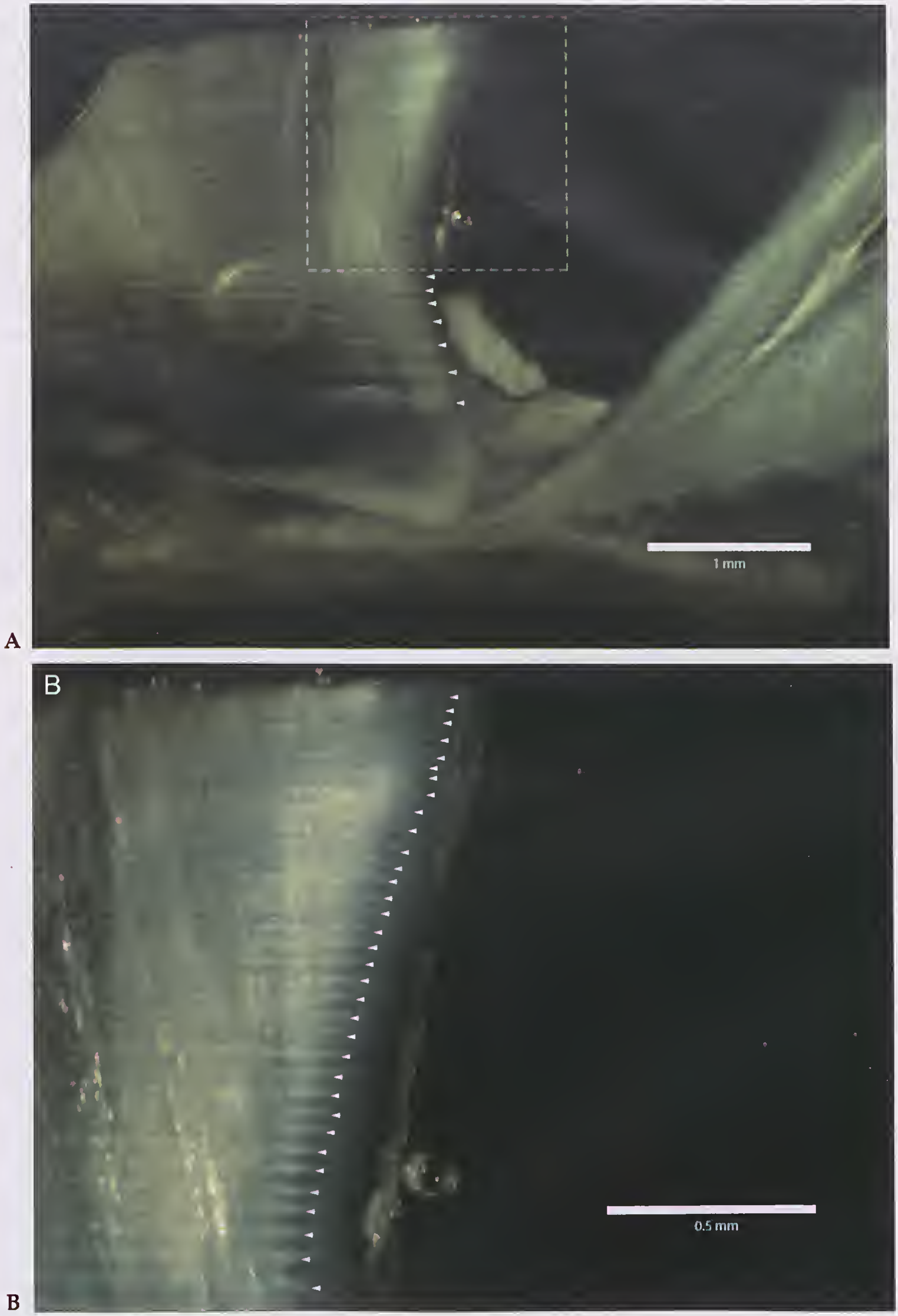

Figure 2. Cross section through sagittal otolith from oldest known Australian snapper highlighting 41 opaque zones. Boxed area in A is magnified in B. Reflected light.

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Eradication of feral cats on Faure Island, Western Australia

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Abstract

The Australian Wildlife Conservancy, lease holders of Faure Island Pastoral Lease, is planning to establish the area as a site for the conservation of threatened mammals and ecotourism. The AWC propose to translocate a suite of native mammal species to the island. To enable the successful recolonization of the island by these species it was necessary to implement a feral cat eradication program.

Feral cat eradication took place in autumn 2001 and comprised two stages – an aerial baiting campaign and a follow up ground baiting, trapping and monitoring program. The baiting campaigns utilized the recently developed feral cat bait (*Eradicat*®) and the toxin sodium monofluoroacetate (1080). Baits were deployed, followed by an intensive 20-day monitoring program. Results from the monitoring program indicated that eradication had been successfully achieved and was accomplished from the baiting application alone. Searches for evidence of cat activity in winter 2001 by an independent team confirmed these results.

Keywords: cat eradication, islands, cat bait, cat trapping, Faure Island.

Introduction

The Australian Wildlife Conservancy (AWC), lease holders of Faure Island Pastoral Lease, plans to establish the area as a site for the conservation of threatened mammals. No native mammals now occur on the island; however there is evidence from sub-fossil deposits that at least four species once occurred on the island. The AWC propose to translocate four native mammal species to the island: the Western Barred Bandicoot (*Perameles bougainville*); Boodie (*Bettongia lesueur*); Shark Bay Mouse (*Pseudomys fieldi*) and Banded Hare Wallaby (*Lagostrophus fasciatus*). To enable the successful recolonization of the island by these species it was necessary to implement a feral cat (*Felis catus*) eradication program. Feral cats became established on Faure Island during the late 19th century and were probably introduced from pearling vessels or by early pastoralists.

The importance of islands to the conservation of Australian mammal species has been well documented (Burbidge & McKenzie 1989; Abbott & Burbidge 1995). One of the key factors in the historic importance of islands has been that most have remained free of introduced predators. Burbidge (1999, pp 72) highlighted the current and future importance of islands to nature conservation and stated that "Australian nature conservation agencies need to pay more attention to the eradication of exotic animals from islands".

Feral cats pose a serious threat to populations of small to medium-sized native vertebrates (Environment Australia 1999; Department of the Environment, Water, Heritage & the Arts 2008). Anecdotal evidence has indicated that predation by feral cats, either acting singly or in concert with other factors, has resulted in the local extinction of a number of species on islands and mainland Australia. Predation by feral cats also affects the continued survival of many native species persisting at low population levels (Dickman 1996; Smith & Quin 1996; Risbey *et al.* 2000) and has prevented the successful re-introduction of species to parts of their former range (Christensen & Burrows 1995; Gibson *et al.* 1995; Priddel & Wheeler 2004). Control of feral cats is recognized as an important conservation issue in Australia today and as a result, a national 'Threat Abatement Plan for Predation by Feral Cats' was developed (Environment Australia 1999; Department of the Environment, Water, Heritage & the Arts 2008). Management of introduced predators is now generally viewed as a critical component of successful reintroduction, recovery or maintenance of small to medium-sized native fauna populations (Christensen & Burrows 1995; Fischer & Lindenmayer 2000).

The Western Australian Department of Environment and Conservation (DEC), through Project 'Western Shield', has been conducting research to develop an effective, broad-scale feral cat control strategy. As a result of our research, a request was made to DEC to conduct a feral cat eradication program on Faure Island. A preliminary study to assess the feasibility of feral cat eradication on the island was conducted in September 2000 (Algar & Angus 2008). During this preliminary study a small-scale trapping program, bait acceptance trial and intensive searches for evidence of cat activity were conducted. Trapping in conjunction with assessment of track activity provides a simple and effective method to assess approximate cat density and distribution (Algar *et al.* 1999). These surveys of feral cat activity indicated there was a population of approximately 40 adult cats on the island at that time (Algar & Angus 2008).

Cats readily consumed non-toxic baits during this preliminary study. On-track bait acceptance trials, using baits placed at 100 m intervals on tracks, indicated that 67 % of the sampled population consumed at least one bait (Algar & Angus 2008). Analysis of cat stomach contents indicated that the principal dietary items being consumed at this time were reptiles, in particular hatchling Gould's Monitors (*Varanus gouldii*). Activity of *Varanus* spp. and their relative size would have provided a reasonable prey source for cats. Bait acceptance by feral cats is, in part, related to the abundance of prey species. Bait consumption generally increases as prey availability declines (Algar *et al.* 2007). On Faure Island, the increased size of young varanids, through summer and into autumn, was likely to reduce their vulnerability to predation. Bait consumption by cats was therefore expected to increase from that recorded in September, given the reduction of this prey resource.

A feral cat eradication campaign was conducted on Faure Island during early autumn 2001, a season when bait acceptance was predicted to be high and the campaign was unlikely to be affected by rainfall. An aerial baiting campaign was adopted as the primary control technique. Following the baiting campaign a ground-baiting, trapping and monitoring program was employed to remove any cats that remained. This control strategy was successfully employed on the Montebello Islands (Algar & Burbidge 2000; Algar *et al.* 2002) and was adopted as a suitable approach on Faure Island. This paper describes the feral cat eradication campaign on Faure Island.

Methods

Site Description

Faure Island is located between 25°48'S and 25°54'S and between 113°51'E and 113°56'E (see Figure 1). The island, an area of 58 km^2, lies within the eastern gulf of Shark Bay at the head of the Hamelin Pool embayment, approximately 18 km east of Monkey Mia. Shark Bay has a semi-arid to arid climate with hot dry summers and mild winters (Wilson 2008). Summer minimum and maximum temperatures average between 20 °C and 35 °C respectively and winter temperatures between 10 °C and 20 °C. Faure Island Pastoral Lease records (1991–2005) show that rainfall falls predominately between March and August with an average of 185 mm per annum (AWC, unpublished data). Most rain falls in the winter months but cyclones bring summer rain some years. There are five major vegetation types on the island: *Acacia* shrublands on an undulating red sandplain which covers the majority of the island: Mallee Eucalypt shrubland; *Spinifex* grassland backing onto the beaches and on the coastal dunes; samphire and *Atriplex* shrubland in and surrounding birridas and saline flats and mangrove communities interspersed with succulent shrublands (Keighery & Muir 2008).

Baits and Baiting Program

DEC researchers at the time of this baiting campaign had recently completed development of a bait to control feral cats. The bait (*Eradicat*®) is similar to a chipolata sausage, approximately 20 g wet-weight, dried to 15 g, blanched and then frozen. The bait is composed of 70 % kangaroo meat mince, 20 % chicken fat and 10 % digest and flavour enhancers that are highly attractive to feral cats (Patent No. AU 781829). The toxic baits were manufactured at the DEC Bait Factory at Harvey and airfreighted to the island. The toxin 1080 (sodium monofluoroacetate) was injected into the baits at a dose of 4.5 mg per bait. Prior to application, the baits were thawed and placed on racks, in direct sunlight. This process allows the oils and digest material to come to the surface of the bait. At this time, all baits were sprayed with an ant deterrent compound (Coopex®) at a concentration of 12.5 gl^{-1} as per the manufacturer's instructions. Ant attack on baits rapidly degrades the bait medium, reducing palatability and the persistence of ants on the bait deters uptake by feral cats (D Algar, pers. obs.). Baits were counted into bags (100 baits per bag) on-site to contain the required number of baits per 1 km^2 baiting cell. Although prior to the adoption of a 'Code of Practice on the Use and Management of 1080' (Health Department, Western Australia) and associated '1080 Baiting Risk Assessments', considerable review and consideration of the hazard the intended baiting could have on known non-target species' populations on the island was undertaken. This analysis concluded that there was unlikely to be any significant effects on non-target species' populations such as Gould's Monitors on the island.

The optimum baiting intensity for feral cats is not currently known and is likely to vary with location, habitat, cat density and time. In an attempt to maximize the availability of baits, a baiting intensity of 100 baits km^{-2} was used. This baiting intensity was aimed at maximizing the likelihood of cats encountering a bait when hungry. The baits were deployed from a Beechcraft Baron E55 aircraft, flying at a nominal speed of 160 kt and 1000 ft (Above Ground Level). The aircraft was guided by an AG-NAV navigation system with pre-set flight lines for the target area. A timing light indicated to the bombardier when baits must be dropped to achieve the desired bait distribution. A flight plan was prepared that ensured flight cells were one km intervals across the island with an additional flight line following the coast, and another deploying baits in the interdunal swales. Baits were distributed on the 28 February 2001.

A ground baiting program was implemented in areas where evidence of fresh cat track activity was observed. Single baits were placed on marked sand pads, swept clean of tracks, at approximately 20 m intervals along tracks or cross-country. The ground baiting program continued on a daily basis until no further evidence of cat activity was recorded.

Trapping Program

An intensive trapping and cat activity monitoring program across the island commenced 10 days following baiting. The 10-day period allowed cats sufficient time to encounter and consume baits prior to any potential bait degradation. Five personnel were involved in the

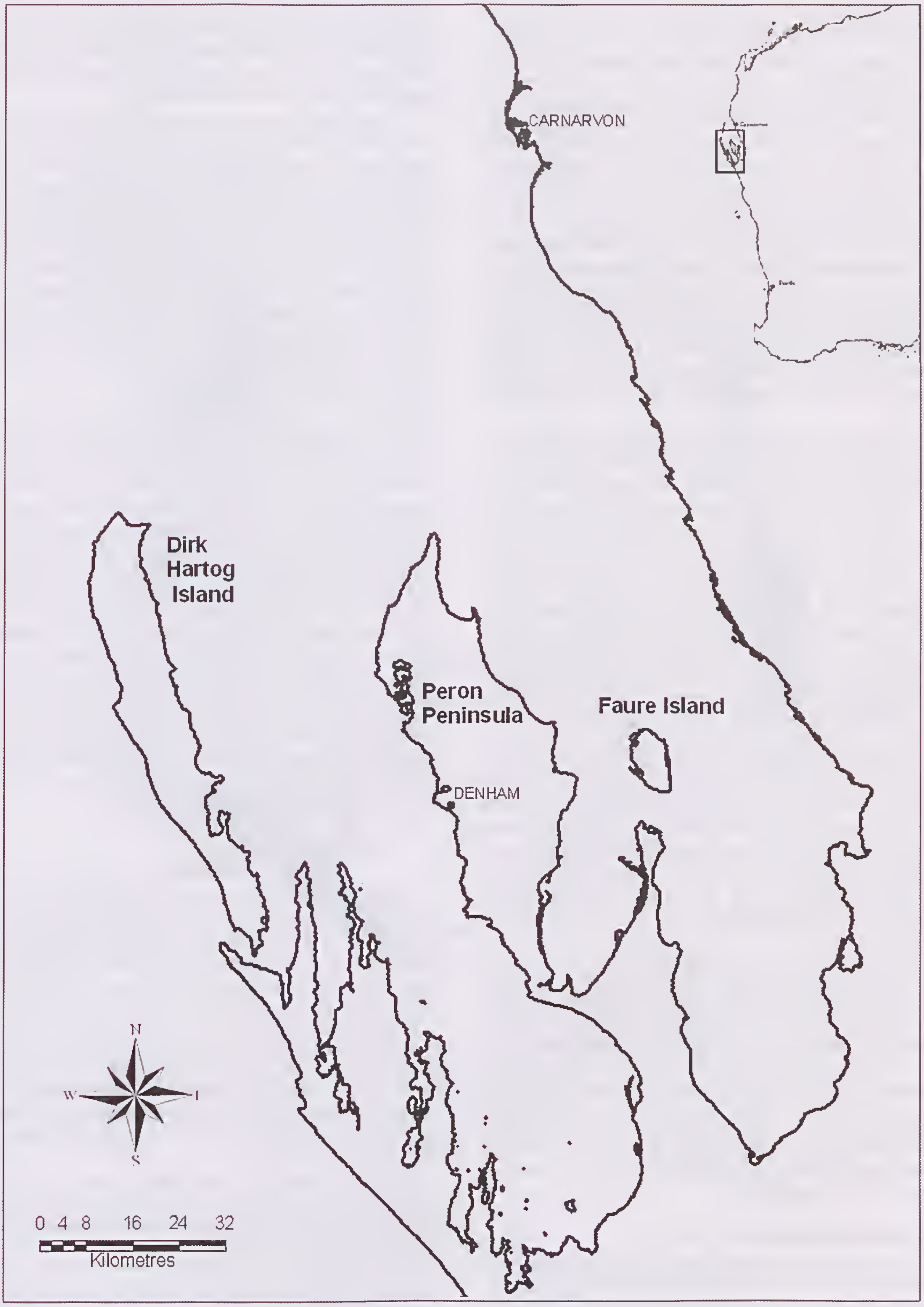

Figure 1. Location of Faure Island.

trapping/monitoring program. The trapping and monitoring exercise was conducted over 20 days in total.

Trapping systems for cats have generally relied on food-based lures as the trap attractant (reviewed by Veitch 1985). A number of other olfactory scents or social odours to entice cats into traps or bait stations have also been used (Veitch 1985; Clapperton *et al.* 1994; Edwards *et al.* 1997). An alternative technique to these systems, using lures that mimic signals employed in communication between cats, has been developed by DEC researchers and proven highly successful. Cats are very inquisitive about other cats in their area; their communication traits are principally reliant on audio and olfactory stimuli. The trapping technique employed padded leg-hold traps, Victor 'Soft Catch'® traps No. 3 (Woodstream Corp., Lititz, Pa.; U.S.A.), a Felid Attracting Phonic (FAP, Patent No. AU 740648, Westcare Industries, Western Australia) that produces a sound of a cat call and a scent lure consisting of a mixture of faeces and urine.

Each trap site consisted of a channel slightly wider than the width of one trap and 80 cm in length, cleared into a bush to create a one-way trap set. Two traps, one in front of the other were positioned at the entrance of the set, at each trap site. A trap bed was made so that when lightly covered with soil, the traps were level with the surrounding ground surface. A guide stick was placed in front of the traps to force animals to lift their foot then push down onto the pressure plate. Both traps were secured in position by a 30 cm length of chain to a 30 cm steel anchor peg. A 12 x 8 x 2 cm foam pad was placed below the pressure plate to prevent soil from falling into the trap bed and compacting under the plate. The traps were then lightly covered with soil.

Cats are lured to the trap set initially by the audio signal produced by the FAP. The FAP is located at the back of the trap set, either concealed under leaf litter or hidden within the bush. The FAP consists of a 36 x 25 mm printed circuit board with a microprocessor data driven voice ROM. As cats approach the trap set they are further enticed into the traps by the scent lure. Approximately 20 ml of this mixture is placed in a shallow depression about 30 cm from the centre of the back trap.

Traps were positioned along all vehicle access tracks and along the coastal/dune areas not readily accessible by motor vehicle. Traps using the FAP/scent lure combination were located at one km intervals and those employing the scent lure alone were positioned at the intervening 500 m intervals. Trap locations were recorded using GPS receivers and are shown in Figure 2. Traps were set in position for a period of at least 11 days before retrieval. In total, 123 trap sets were placed over the island during the trapping period. Ninety four of these were in place for more than 14 days to achieve a total of 1,819 trap nights.

Monitoring Program

The monitoring program consisted of recording the presence or absence of cat track activity along transects, trap locations and sand plots. Four methods were used to monitor for cat activity on the island: observation from the vehicle undertaking the trapping program along access tracks; cross-country transects conducted both on-foot and by motor cycle; and intensive on-foot searches along the beaches, interdunal swales and mangroves. The location of cat track activity was recorded and the area swept clear following inspection.

There was a 59 km network of vehicle access tracks on Faure Island. These tracks were inspected daily for evidence of cat activity from a 4WD vehicle, driven at a speed less than 10 kmh^{-1}. On-foot searches, undertaken every second day, were conducted from the Fishing Hole to the Point on the east coast, the entire south coast and much of the west coast, providing 22 km of walking transect (see Figure 2). The beach areas north of the Fishing Hole on the eastern and northern coasts were inter-tidal and comprehensively covered by the vehicle transect. Further to the regular inspection of transects; intensive, random searches were conducted routinely every two days of dune and mangrove areas, not otherwise readily accessible (Figure 3).

Motorcycle transects were established such that they bisected each block of inaccessible land, along the longest axis (Figure 2). The end-points of each transect were measured from a geo-referenced map and stored as waypoints in a GPS receiver. Sand plots were placed at nominal intervals of one km, along each transect. Each sand plot contained a FAP attractant, with loud speaker fitted, hidden within the foliage of a shrub. The sand plot was smoothed over the entire circumference of the shrub. The total area of the sand plot varied with the size of the shrub but was at least two metres wide. All transects were traversed prior to sand plot placement and any sign of cat track activity was removed from approximately 10 m either side of the alignment. Following sand plot placement, transects were traversed every second day, for at least nine days, at a speed of less than 10 kmh^{-1}. Evidence of cat track activity was also searched for while traversing the motorcycle transects. A total of 21 sand plots were placed along 28 km of transect.

Results

Evidence of fresh cat activity was observed around the water points, homestead and Landing by the caretaker manager prior to the baiting program on a visit to the island in mid February 2001 (Dick Hoult pers. comm.). Following the aerial baiting program no cats were trapped and no sign of fresh cat activity was observed at any of the trap locations or sand plots over the entire trapping/monitoring period. The only evidence of fresh cat activity was in the area bounded by the Landing, Landing Rd., Buffel Rd. and the coast. Activity was recorded along the 4WD transect on Landing Rd., Buffel Rd. and along both the on-foot and motorcycle transects that traverse this block.

Intensive searches identified a core of activity immediately inland of the coastal dunes and indicated that two to three cats had survived the aerial baiting program. This activity was recorded on the first day of monitoring and a localized ground baiting was conducted in the area, that day and the following. Subsequent assessment of cat tracks and activity in the ground-baited area over the next two days indicated that at least five baits were positively taken by cats. Transects

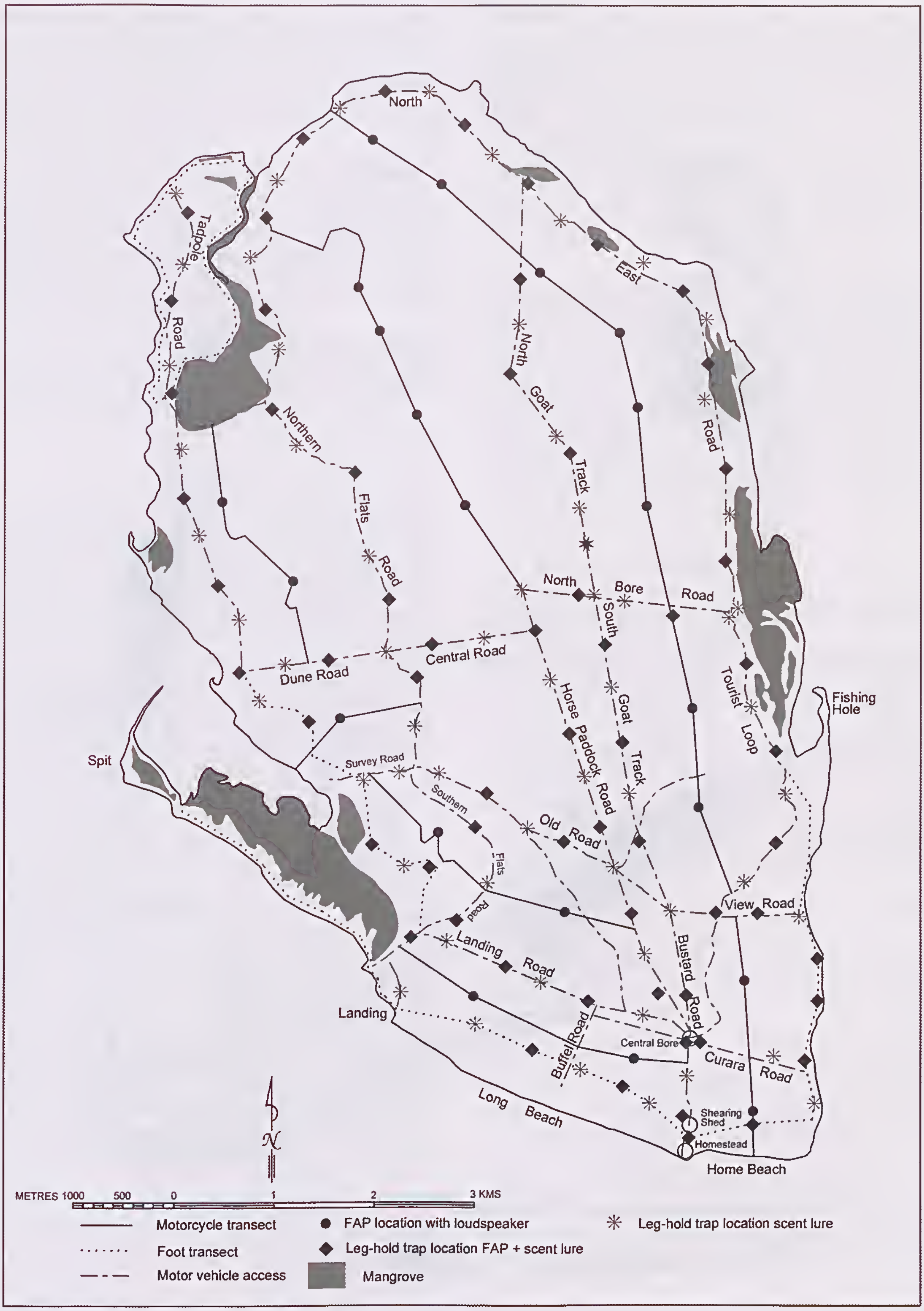

Figure 2. Locations of trap sets, monitoring plots, and transects.

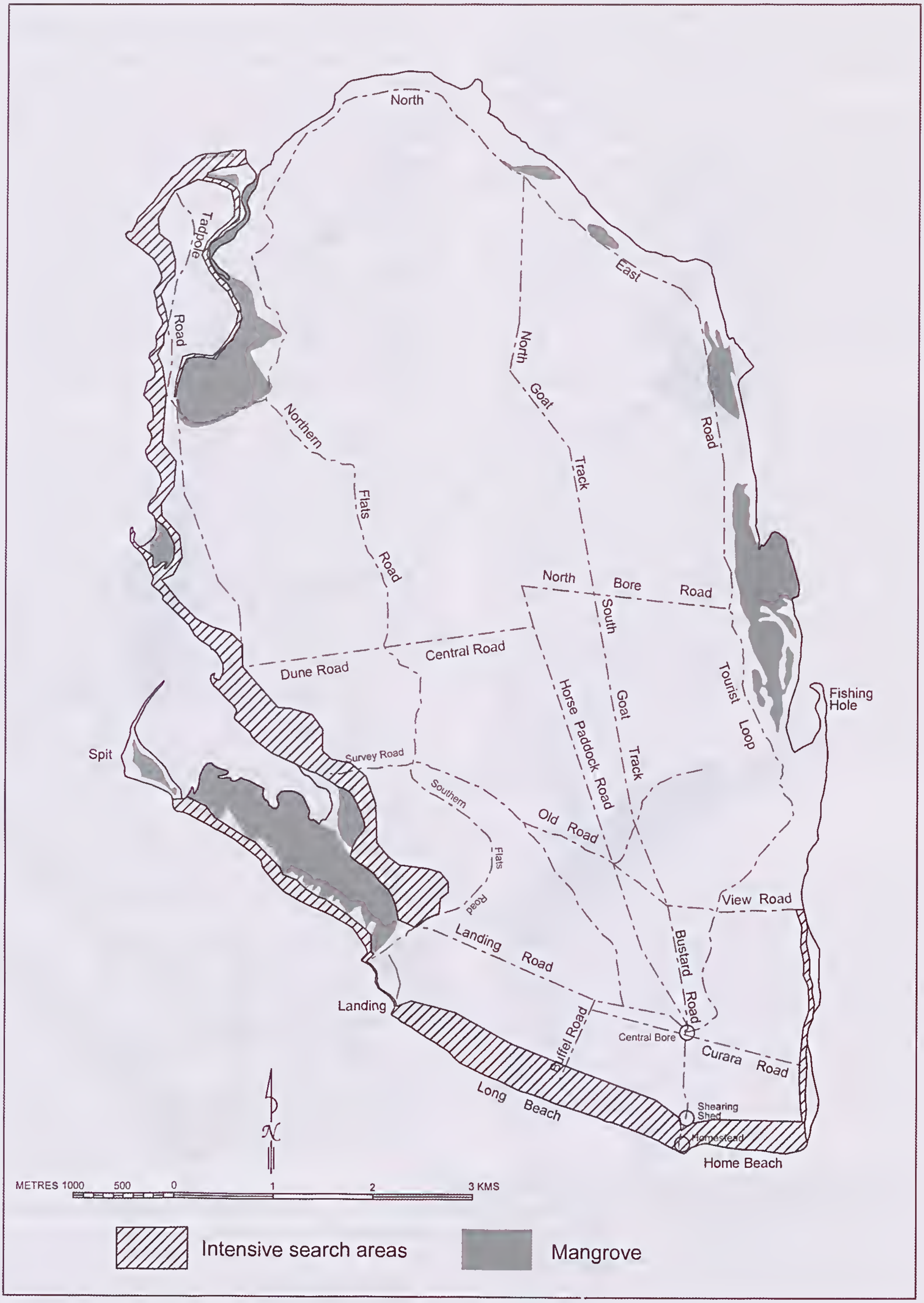

Figure 3. Location of intensive, random searches conducted in dune and mangrove areas.

and sand plots in the area were inspected every second day; however no further fresh cat activity was found over the following 16-day monitoring period. Intensive searches in this area failed to locate any baits. Elsewhere on the island, baits were regularly found during searches, and it is believed that no baits were aerially distributed over the area.

The aerial baiting campaign was responsible for the removal of at least 90 % of the cats on the island, given that a population of approximately 40 adult cats was present on the island the previous spring and that only two to three cats survived the aerial baiting program. This figure may have been higher as natal recruitment into the population would have occurred over the late spring and summer.

An intensive survey for evidence of cat activity conducted in June 2001 by an independent team confirmed that eradication of cats on the island had been successful (Thomas & Whisson 2001).

Discussion

Cat eradication programs on islands are usually conducted using a combination of baiting, trapping and hunting (Rauzon 1985; Veitch 1985; Bloomer & Bester 1992; Bester *et al.* 2000; Rauzon *et al.* 2008). These eradication programs have met with varied success, their success and time to completion having been limited in part by lack of an effective bait and trap lures.

It appears that eradication was achieved at Faure Island and the failure to locate any cat activity with such an intensive monitoring program strongly supports this. However, only time and subsequent inspections will provide absolute confirmation. Genetic material gathered during the initial feasibility study will indicate whether any cat found on the island in the future is a remnant of the original population or a subsequent introduction.

This aerial baiting program was conducted prior to equipment being available to download maps of bait distribution from the aircraft. Had a bait distribution map been available, any area where baits had not been deployed could have been flown again. The fact that the remaining cat/s readily took baits, complete coverage of the island with the prescribed baiting density may have resulted in total eradication from the aerial baiting program alone. Nowadays, mapping of bait distribution is undertaken and these are provided following all baiting programs. Monitoring cat activity and providing indices of abundance immediately prior to baiting has also been implemented to provide a more accurate measure of the impact of the baiting exercise.

Given the data in Nogales *et al.* (2004), Faure Island may be the third largest island in the world where feral cats have successfully been eradicated. The cat eradication campaign and monitoring program on Faure Island was achieved within a month. Elsewhere in the world, cat eradication projects on islands have often taken months or years, or are still ongoing (Nogales *et al.* 2004). It is difficult to compare the efficacy of the eradication campaign conducted on Faure Island with others, which have taken place on islands of different climate, terrain, shape and size and with different prey availability. However, the advances in cat control strategies developed by DEC may provide the techniques necessary for effective eradication of feral cats from many islands around the world. Further feral cat eradication programs for a number of other islands off the Western Australian coast are now being planned.

The four threatened native mammal species: the Western Barred Bandicoot; Boodie; Shark Bay Mouse and Banded Hare Wallaby proposed for translocation to Faure Island have since successfully recolonized the island following eradication of cats (Richards 2007).

Acknowledgements: We gratefully acknowledge the AWC for providing the funds to conduct this eradication campaign and the assistance of Brett Chambers in logistical support. Special thanks go to Dick Hoult whose knowledge and assistance on the island proved invaluable, to his wife, Margaret for the never ending supply of cakes and biscuits and putting up with Dick's absence from home and to his son, Dennis for transporting the equipment to and from the island and his sense of humour while manhandling the gear. We would also like to thank Neil Thomas and Leigh Whisson for undertaking the survey to confirm that cats had been eradicated from the island. We thank J. Richards (formerly with AWC) and two anonymous referees for comments on an earlier draft. Figure 1 was produced by S. Hilmer and Figures 2 and 3 by R. Doria. The techniques used in this program have been approved by DEC's Animal Ethics Committee.

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Opportunistic observation of predation of Loggerhead turtle hatchlings by feral cats on Dirk Hartog Island, Western Australia

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Abstract

This note reports the opportunistic observation of predation of Loggerhead turtle hatchlings at Turtle Bay on Dirk Hartog Island, Western Australia. Data were collected on the movement patterns of three feral cats, fitted with GPS data-logger/radio-telemetry collars in the vicinity of turtle nests. This data as well as field observations demonstrate a distinct patrolling of Turtle Bay and surrounding areas. Furthermore, predation of turtle hatchlings by cats was confirmed by their presence in the stomach contents of one of the cats.

Keywords: feral cat, Loggerhead turtles, predation, Dirk Hartog Island

Introduction

Globally, Loggerhead turtles (*Caretta caretta*) inhabit tropical, sub-tropical and temperate marine waters (Bolten and Witherington 2003). Within Australia, they are found in the eastern, northern and western waters of the continent (Prince 1994; Limpus 2008). Loggerhead turtles are listed as 'Endangered' under the Commonwealth *Environment Protection and Biodiversity Conservation Act* 1999 and the IUCN Red List (IUCN 2009).

Predation by feral cats is known to have a deleterious impact on endemic land vertebrates and breeding bird populations on both offshore and oceanic islands around the world (van Aarde 1980; Moors & Atkinson 1984; King 1985; Veitch 1985; Bloomer & Bester 1992; Bester *et al.* 2002; Keitt *et al.* 2002; Blackburn *et al.* 2004; Martinez-Gomez & Jacobsen 2004; Nogales *et al.* 2004). Predation by feral cats on Green turtle hatchlings (*Chelonia mydas*) on Aldabra Atoll, Seychelles has also been demonstrated (Seabrook 1989), although the impact could not be fully determined.

Turtle Bay and environs of Dirk Hartog Island is a major nesting ground for Loggerhead turtles with 800–1500 females breeding each year between November and March (DEWHA 2010). Footprints of feral cats around turtle nests were observed during a pilot study to assess the efficacy of a baiting program to eradicate feral cats on the island (Algar *et al.* submitted).

Materials and Methods

In March 2009, sixteen feral cats were trapped on the island for a pilot study to assess bait efficacy. Two of these cats (DH27 and DH27.2) were caught near Turtle Bay while another, MB8 was trapped close to the west coast (Figure 1). The cats were fitted with a GPS data-logger/radio-telemetry collars (small Feline/Possum GPS collar, weight: 105 g, dimensions: 55mm wide x 34 mm deep by 37 mm high under neck, Sirtrack Ltd, New Zealand), which collected and stored a location point every 40 minutes (DH27, DH27.2) and 10 minutes respectively (MB8). Cats were released at site of capture and not disturbed during the period of data collection.

A feral cat baiting program was conducted four weeks after the release of the cats (Johnston *et al.* 2009; Algar *et al.* submitted). Collars were retrieved following the death of individual cats, including DH27.2, following baiting. Cats DH27 and MB8 were shot at the conclusion of the study as they did not consume a toxic bait. No stomach analysis was conducted.

A 30 m section of beach to the west of Cape Inscription that encompassed an active turtle nest was inspected daily over 5 days for fresh feral cat activity. Observations and interpretations were made from the footprints made by cat(s), turtles, crabs and other species. Prints were cleared daily by sweeping the 'study area' with a broom.

Results

Each of these cats was male and weighed; DH27, 5.1 kg, DH27.2, 4.5 kg and MB8, 5.5 kg when trapped. A compilation of the location data derived from GPS data-logger collars, between 30 March and 30 April 2009, is shown in Figure 2. The data indicate almost linear walking patterns as the cats foraged along the beach. In addition, fresh cat tracks, thought to be MB8, were seen daily at the turtle nesting sites as shown in Figure 3. Toxicosis following ingestion of the poison '1080' (sodium monofluoroacetate) used in the feral cat baiting program often results in regurgitation of stomach contents. Vomitus of an entire turtle hatchling and

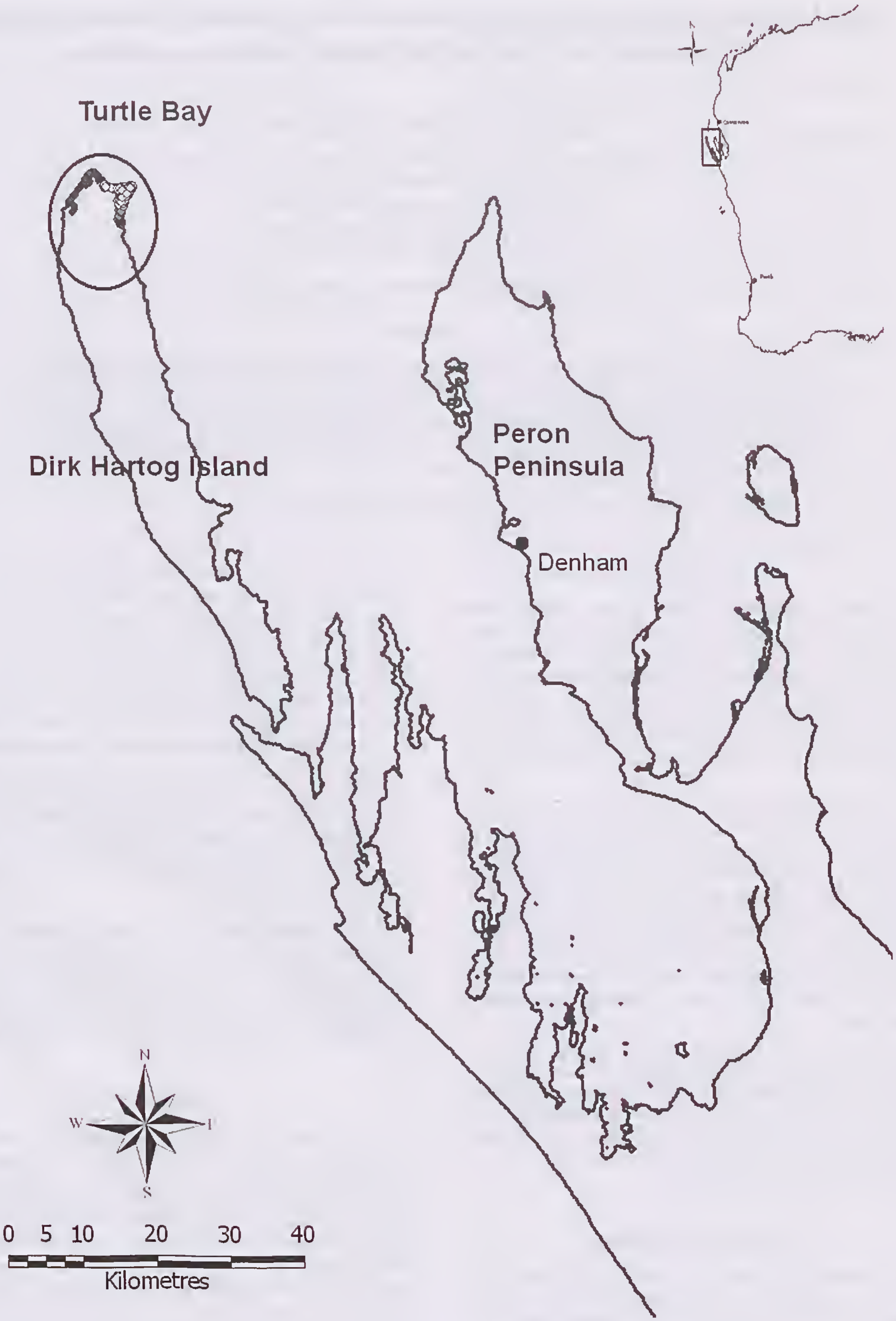

Figure 1. Location map of study area, Turtle Bay on Dirk Hartog Island (circled area).

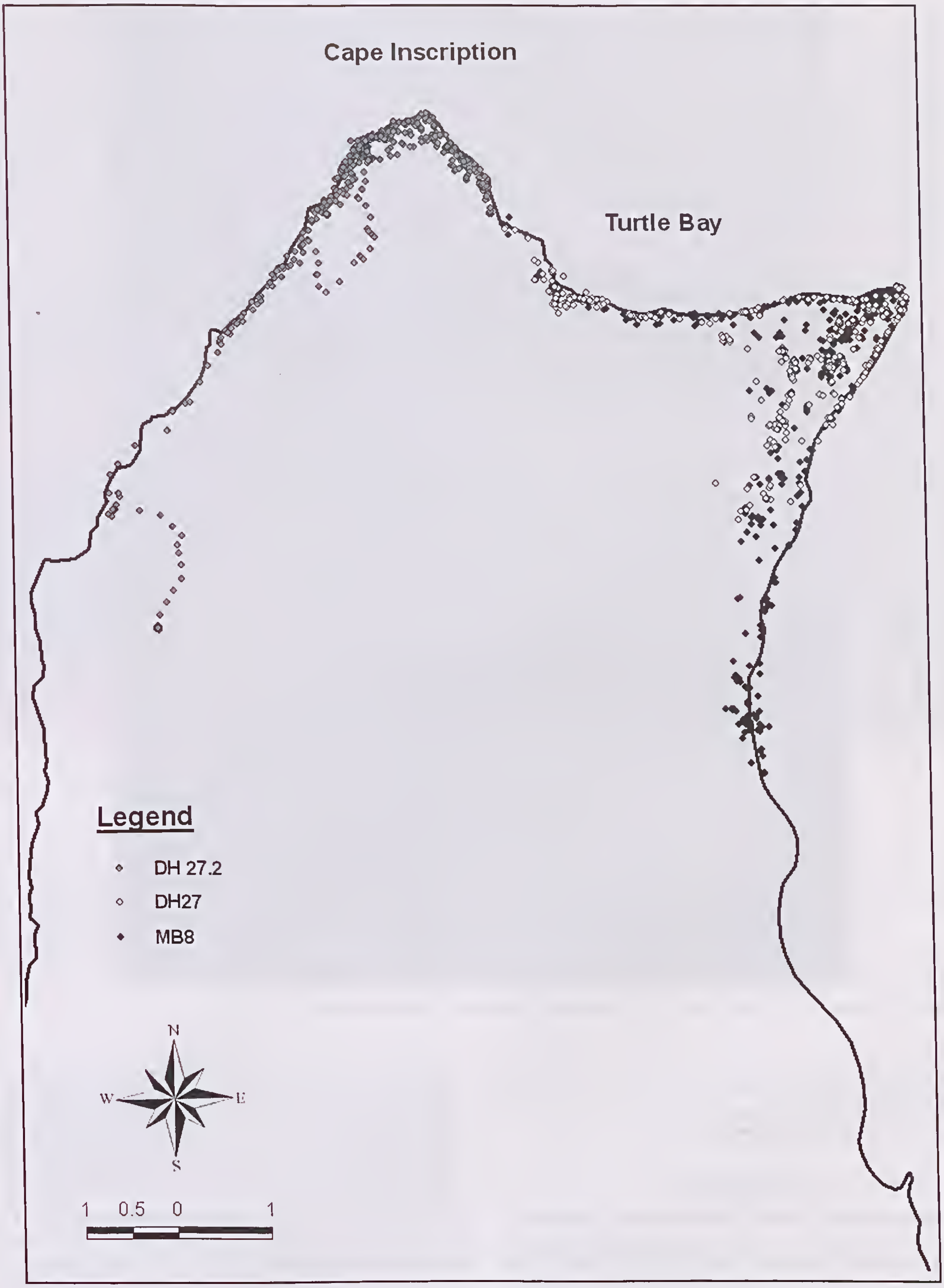

Figure 2. GPS-locations of three feral cats at and near Turtle Bay collected over a period of 10 days (MB8, 29.3-8.4.2009) and 30 days (DH27, DH27.2, 30.3- 30.4.2009).

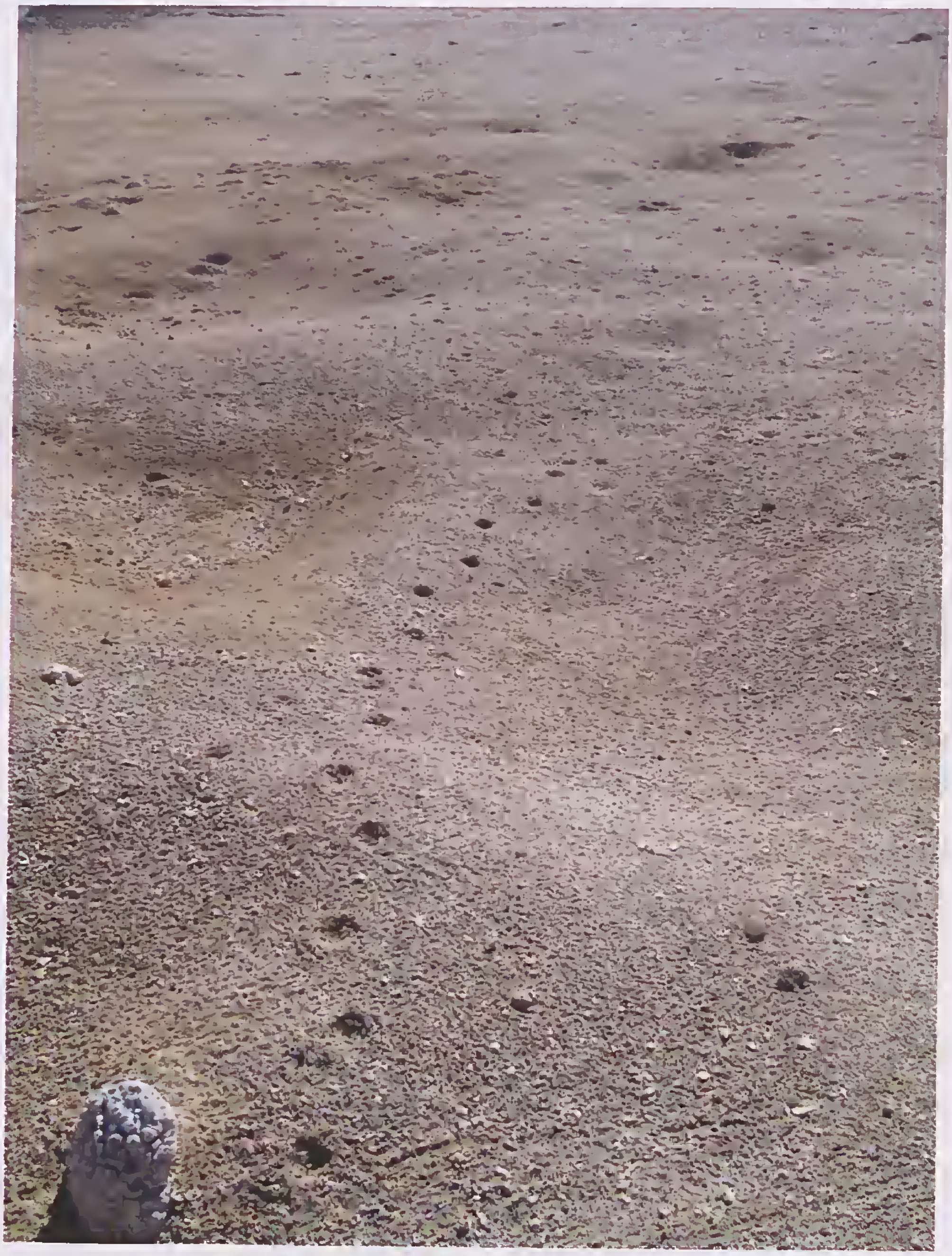

Figure 3. Cat tracks on Turtle Bay beach next to Loggerhead Turtle nests (left from cat tracks).

unidentified rodent was located within 10 metres of the carcass of DH27.2 (see Figure 4).

Discussion

Many terrestrial species, both native and introduced, are known to predate turtle eggs or emergent hatchlings (Limpus 1973; Environment Australia 2003; Limpus 2008) however, predation by feral cats has rarely been demonstrated (Seabrook 1989). Hamann *et al.* (2006) suggested that predation by feral cats on Flatback turtles (*Natator depressus*) may be of management concern on West Island in the Sir Edward Pellew group, Northern Territory however, despite the presence of many cat tracks at nest sites, predation by cats was not observed. The finding reported in this note, despite a small data set, confirms that feral cats, known to be opportunistic hunters (Jones & Coman 1981; Dickman 1996; Paltridge *et al.* 1997; Risbey *et al.* 1999), predate Loggerhead turtle hatchlings and provides further confirmation that feral cats will consume large prey items (Marks *et al.* 2006; Hetherington *et al.* 2007). The regurgitated prey items, (Figure 4) were entire and stained with the Rhodamine B dye used in the poison baits. The extent of the predation by feral cats on Dirk Hartog Island on turtle hatchlings is not known, but a planned feral cat eradication program on the island, proposed to commence in 2011 may improve hatching survival of Loggerhead turtles at Turtle Bay.

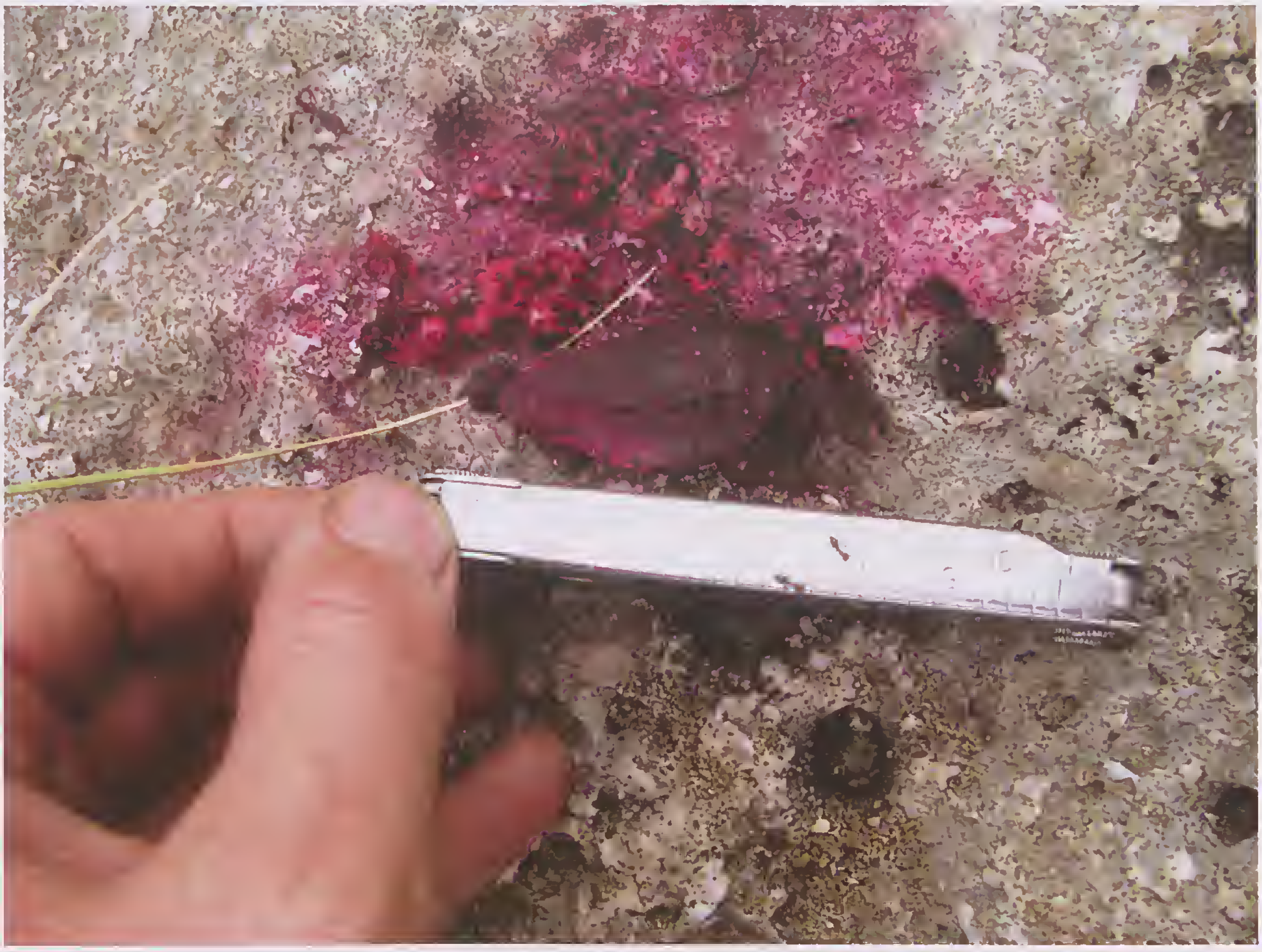

Figure 4. Regurgitated Loggerhead Turtle hatchling found near cat DH27. 2 (Note: red dye is Rhodamine B that was incorporated into the baits).

Acknowledgments: Additional field assistance was provided by Mike Onus, Neil Hamilton, Bruce Withnell, Katrin Koch, David Hawke and Steve Virgin of the WA Department of Conservation and Environment (DEC). Logistic support was provided by the staff at the DEC Denham office especially Brett Fitzgerald. Advice regarding Loggerhead turtle biology was sourced from Bob Prince, Linda Reinhold and Sabrina Trocini.

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Journal of the Royal Society of Western Australia, 93: 147–151, 2010

Invasive potential of a South-American fish species, *Geophagus brasiliensis*, in the Swan River, Western Australia: based on tolerance to instantaneous and gradual changes in salinity

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Abstract

The south-west of Western Australia is a biodiversity hotspot and has a high proportion of endemic freshwater fishes. None of the native fish species are primary piscivores and with the exception of the freshwater cobbler (*Tandanus bostocki*) all species are small (< 200 mm total length (TL)).

The introduction of non-native freshwater fish species is considered one of the most damaging threats to this region's native fish diversity. Recently a new, large (maximum size 300 mm total length) feral fish species, the South American Pearl Cichlid (*Geophagus brasiliensis*), was found in Bennett Brook, a small tributary of the Swan River.

The aim of this study was to determine the salinity tolerance of the feral Pearl Cichlid in order to predict it's invasive potential in watercourses of the Swan River catchment. *Geophagus brasiliensis* tolerated direct transfer from fresh water to 18–27 PPT with no mortality but more importantly, *G. brasiliensis* was able to resist gradual transference from freshwater to salt water (36 PPT) with very low mortalities. Therefore, ongoing control efforts are required in order to limit/prevent the invasion of this feral species into the Swan and other catchments in the southwest of Western Australia.

Keywords: aquatic biodiversity, freshwater fish, introduced species

Introduction

Biodiversity 'hotspots' are areas of exceptional concentrations of endemic species undergoing exceptional loss of habitat (Myers *et al.* 2000). Of the 34 currently recognized biodiversity hotspots in the world only one is situated in Australia; the south-west corner of Western Australia. The south-west of Western Australia has the highest proportion of endemic freshwater fishes (80%) of all of the major Australian Drainage Divisions. Ten species of native freshwater fish occur in south-western Australia, eight of which are endemic to the region (Morgan *et al.* 1998; Allen *et al.* 2002). None of the native fish species are large piscivorous predators; only the Freshwater Cobbler (*Tandanus bostocki*) typically attains a maximum size greater than 200 mm total length (TL) (Morgan *et al.* 1998; Allen *et al.* 2002).

Invasions of non-native species are increasingly recognised as one of the most damaging threats to biodiversity, especially the introduction of freshwater fish species (Kolar & Lodge 2002). At present nine species of non-native fish (Rainbow Trout, *Oncorhynchus mykiss*; Brown Trout, *Salmo trutta*; Redfin Perch, *Perca fluviatilus*; Silver Perch, *Bidyanus bidyanus*; Eastern Mosquitofish, *Gambusia holbrooki*; One-spot Livebearer, *Phalloceros caudimaculatus*; Goldfish *Carassius auratus*; Carp, *Cyprinus carpio*; Rosy Barb, *Puntius conchonius*) are found in the south-west of Western Australia (Morgan *et al.* 2004; Maddern 2008; DL Morgan and SJ Beatty personal communications). Human-mediated translocation [*e.g.* ongoing stocking of trout species for recreational fishing; biological control (Eastern Mosquitofish); aquaculture escapees (Silver Perch); ornamental escapes (Goldfish, Koi Carp, Rosy Barb)], either deliberate or accidental, is the major vector of introduction (Morgan *et al.* 2004).

Recently a new feral species (ornamental escape), the South American Pearl Cichlid (*Geophagus brasiliensis*) (Figure 1), was recorded in Bennett Brook (Figure 2), a tributary of the Swan River. The coastal drainages of Uruguay (Rio da Prata) and south-eastern Brazil (Amazon Basin) are the natural habitat of *G. brasiliensis*, a secondary-division freshwater fish (Axelrod & Schultz 1955; Lowe-McConnell 1991). Primary-division freshwater families are strictly confined to freshwater, whereas secondary-division freshwater families are generally restricted to freshwater but may occasionally enter salt water. *Geophagus brasiliensis* is known to occur in brackish water (14 PPT; Mazzoni & Iglesias-Rios 2002)

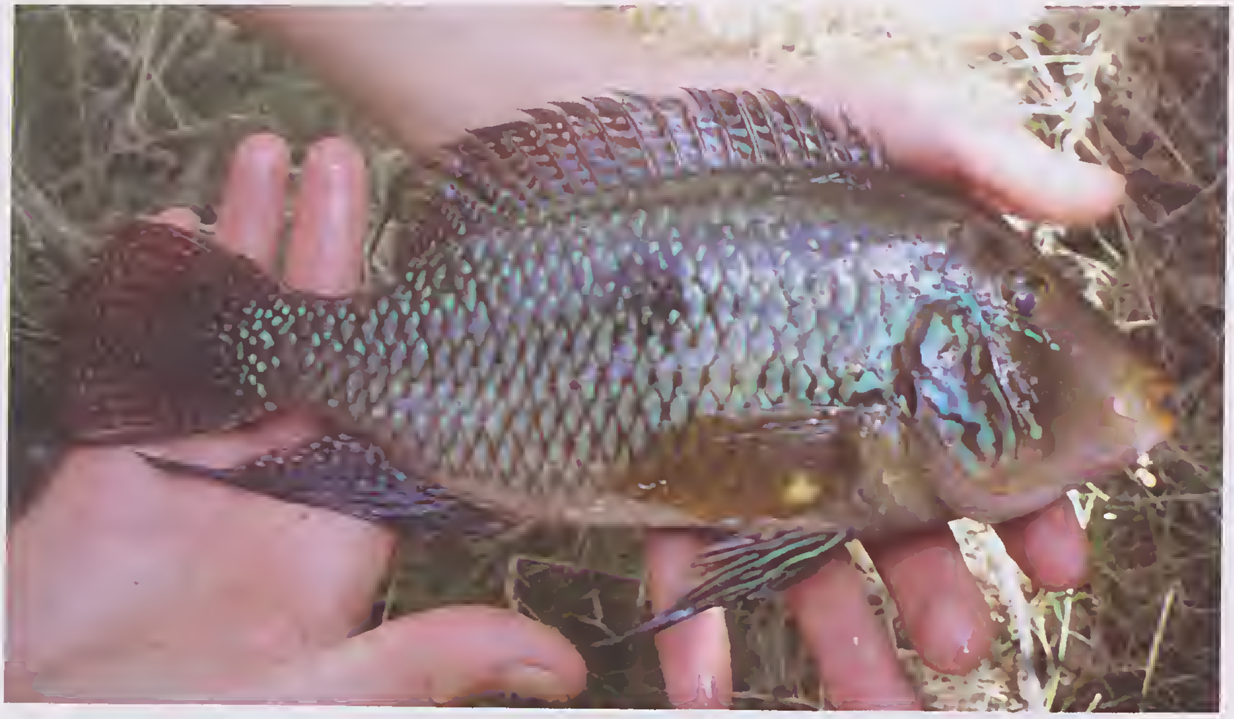

Figure 1. Pearl Cichlid, *Geophagus brasiliensis*, captured in Bennet Brook.

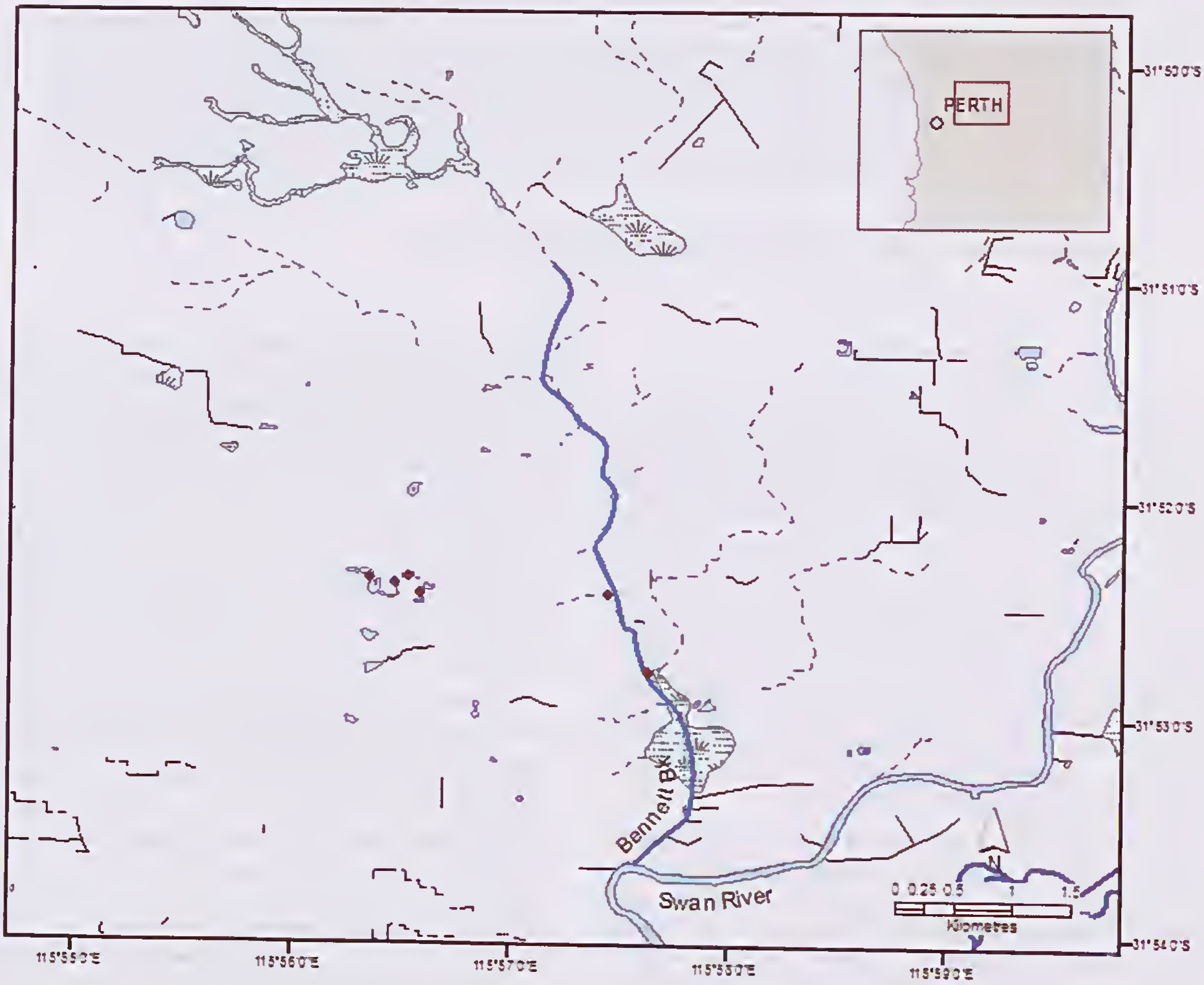

Figure 2. Location of Bennet brook in the Swan River Catchment. Red diamond markers indicate locations where *G. brasiliensis* has been found.

and it may be expected that this species would tolerate elevated levels of salinity.

The Swan River at the confluence with Bennett Brook is brackish for most of the year (Figure 2). Therefore, this species may be able to tolerate salinities that occur in the Swan River. The aim of this study was to determine the salinity tolerance of this population of Pearl Cichlid in order to predict its invasive potential in watercourses of the Swan River catchment.

Methods

Instantaneous Salinity Increase Tolerance Tests

Experimental 72 L aquaria (0.3 x 0.4 x 0.6 m) were partitioned in five equal sections using perforated plastic sheets. Pearl cichlids were captured from Bennett Brook using seine nets and placed individually in a section within experimental aquaria. Each aquarium was aerated and 25% of its volume replaced three times per week. Fish were fed commercial fish pellets twice a week. Fish were acclimatised for two weeks in the experimental aquaria prior to being subjected to salinity tolerance trials.

The first trial was conducted at a water temperature of ~15 °C (avg. 14.8 °C, 0.62 95% CI) using 60 fish (avg. 67.6 mm Standard Length [SL], 4.3 95% CI) divided over 12 experimental tanks. Treatments for the first trial consisted of 0 (control), 4.5, 9, 18 and 36 PPT. Aquaria were randomly assigned to each treatment and mortality was recorded daily.

The second trial was conducted at a water temperature of ~21°C (20.7 °C, 0.16 95% CI) using 64 fish (86.5 mm SL, 2.7 95% CI) divided over 13 experimental aquaria. Treatments for the second trial consisted of 0 (control), 4.5, 9, 18, 27 and 36 PPT. Aquaria were randomly assigned to each treatment and mortality was recorded daily.

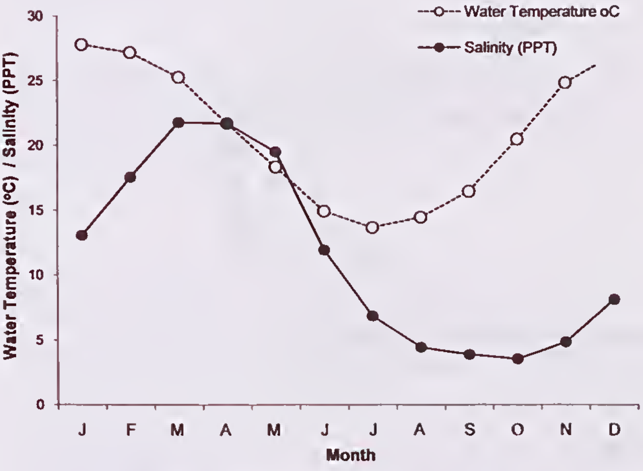

Figure 3. Monthly changes in salinity (PPT) and water temperature (°C) of the Swan River near the mouth of Bennett Brook between 1995–2007 (data source Swan River Trust).

Progressive Salinity Increase Tolerance Test

Fish were captured in the wild using seine nets and acclimatized to laboratory conditions in the 72 L tanks described above. After two weeks, 18 fish (avg. 92.6 mm SL, 5.5 95% CI) were individually placed in 7 L containers. Six containers were randomly assigned as controls (0 PPT), while salinity in the remaining 12 containers was gradually increased from 0 to 36 PPT over a nine-day period (4.5 PPT per day). After the initial nine day period, salinity levels were retained at 36 PPT for an additional 15 days. Water in all 18 containers was replaced daily and fish were fed commercial pellets twice a week. The surviving fish in 36 PPT were all transferred to one large tank (0.5 x 0. 5 x 1.2 m) for a further 26 days before the experiment was concluded.

Results

In both instantaneous salinity increase experiments (Figures 4A,B), no mortality was observed up to 27 PPT. A direct transfer to 36 PPT caused mortality of all fish within 24 hours at 14°C and within six days at 21°C. In contrast, low mortality (< 10%) among specimens of *G. brasiliensis* was observed when salinity was gradually increased over a nine day period (Figure 4C). The majority of fish survived at 36 PPT for a period of 41 days.

Discussion

Geophagus brasiliensis from Bennett Brook tolerated direct transfer from fresh water to 18–27 PPT with no mortality (Figures 4A,B). Fish transfer directly to seawater (36 PPT) did not survive in both instantaneous trials but when conducted at 15°C most fish died within 24 hours while in water of 21°C several individuals survived for six days. Compared to temperate fish species the neotropical *G. brasiliensis* has a low thermal tolerance (Tantin & Petersen 1985) and at 15°C the fish most likely suffer from both salinity and temperature stress.

More importantly, this study clearly demonstrated that *G. brasiliensis* from Bennet Brook is able to resist gradual transference from fresh water to sea water and survive in sea water for a long period of time, *i.e.* up to at least 40 days) with very low mortalities (Figure 4C).

Based on the results of this study it is highly unlikely that without effective management action, the feral *G. brasiliensis* will be confined to Bennett Brook. *Geophagus brasiliensis* is likely to spread throughout the interconnected watercourses of the upper Swan catchment. It is, however, unclear whether *G. brasiliensis* would be able leave the Swan system and to migrate along the coast and invade other river systems in south-western WA. The spread of *G. brasiliensis* may pose a threat to aquatic biodiversity for several reasons. In the first place, the relatively large *G. brasiliensis* is territorial and aggressive towards conspecifics and other fish species especially during the breeding season. Secondly, *G. brasiliensis* is an omnivore (de Moraes *et al.* 2004; Figure 5, de Graaf unpublished data) and will compete with most native fish species for the same food resources (Morgan *et al.* 1998). Furthermore, introductions of *G.*

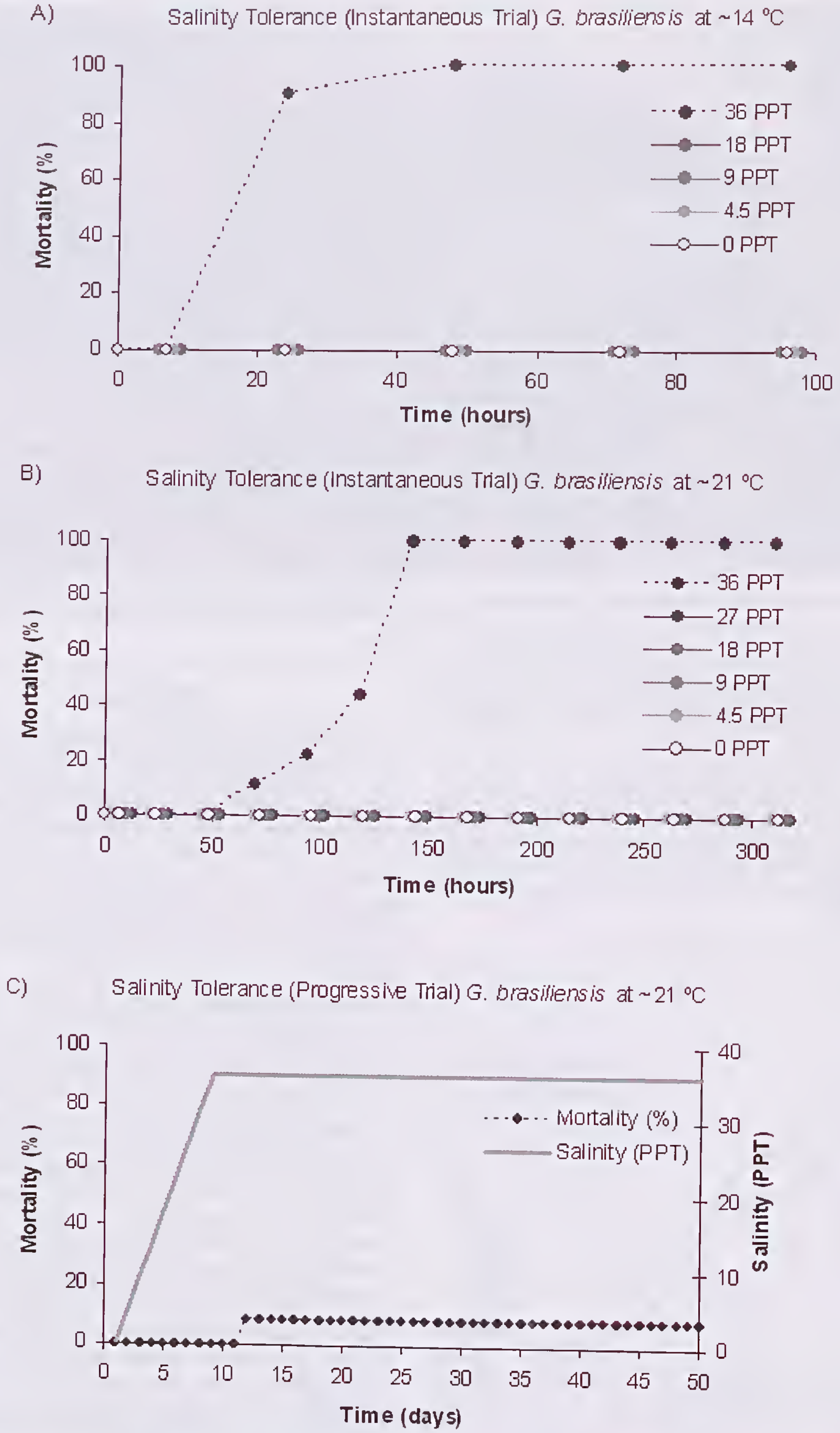

Figure 4. Percentage mortality of *Geophagus brasiliensis* in the different salinity treatments in the instantaneous salinity increase tolerance trails at (A) 14 °C, and (B) 21°C and (C) the percentage mortality of *Geophagus brasiliensis* recorded during the progressive salinity increase tolerance trial.

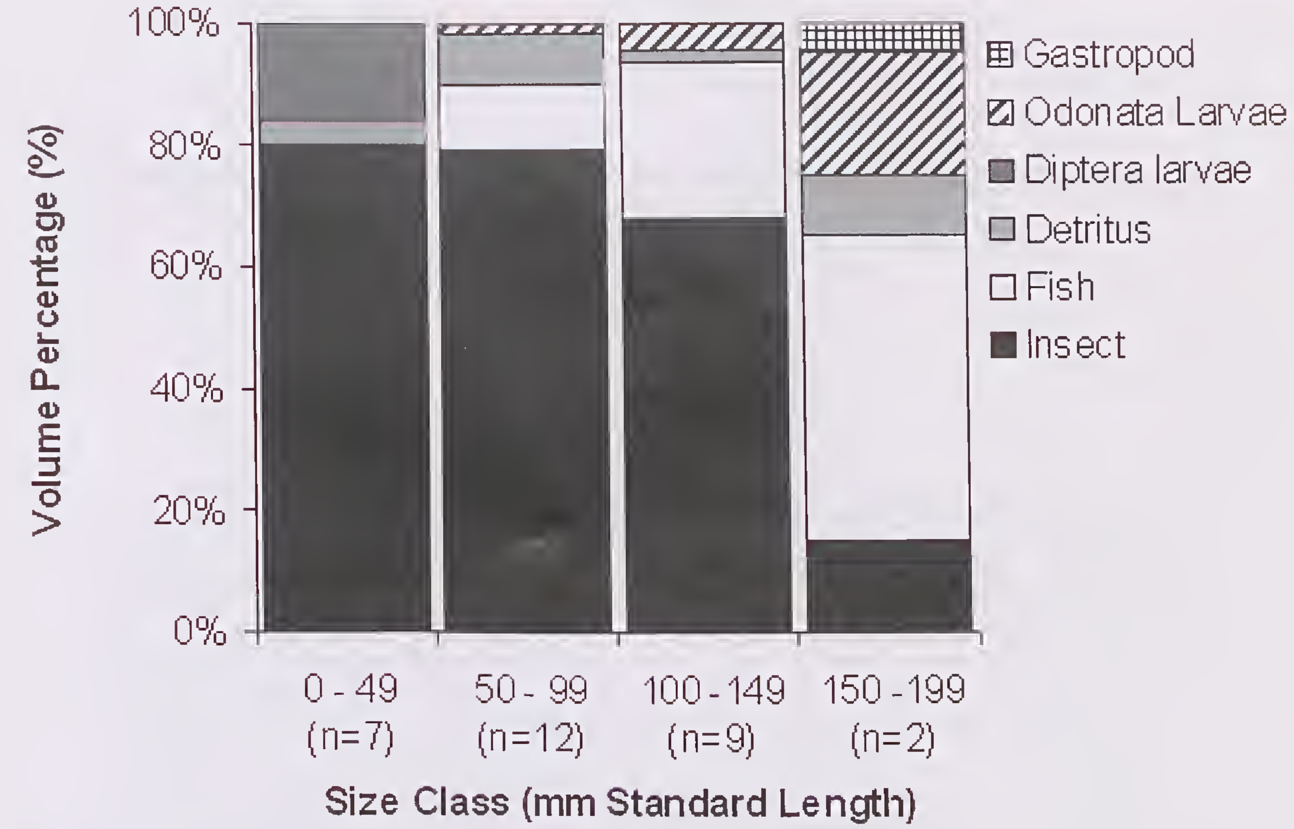

Figure 5. Ontogenetic changes in the diet of *Geophagus brasiliensis* at Bennet Brook (May 2006).

brasiliensis have been reported in Taiwan, USA, Philippines (www.fishbase.org) and the Tweed River, New South Wales, Australia.

Future research should focus on a) the reproductive biology to determine size-at-maturity, breeding period and its potential ability to reproduce in the Swan River, and b) the diet of *G. brasiliensis* to determine the potential impact on native fish through competition and/or predation. The results of this preliminary study clearly suggest that ongoing control efforts are required in order to limit/prevent the invasion of the species into the Swan catchment.

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Presentation by Emeritus Professor Don Bradshaw
The Royal Society of Western Australia Medallist 2010

Ecophysiology and conservation of wildlife in Western Australia

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Abstract

Ecophysiology is the study of the relevance of physiological processes in both plants and animals to ecological constraints imposed by their environment. My own research has focused on Australian animals, specifically reptiles, amphibians, birds and marsupials, but lately, plants have become important in working with the nectarivorous Honey possum. The discipline of physiology was created by the Frenchman Claude Bernard in the 19th century and he dealt with animals only as models for the human condition. It was only in the 20th century that physiologists began to study animals in their own right, not just as proxies for humans, and the developing discipline of comparative physiology laid the grounds for the emergence of ecophysiology. Animals adapt to changes in their environment, or they perish, and we are faced with an ever-lengthening list of species that are now threatened with extinction in Australia. A long-term study of the ecophysiology of the unique suite of native animals living on Barrow Island, for example, has revealed the extent to which they are amazingly adapted to survive in this difficult arid environment, but periods of stress have also been identified in some species when their ability to maintain physiological homeostasis is compromised. Our knowledge of the ecophysiology of the Western Australian fauna is limited to only a very small number of species but, in each case, it empowers us to identify those elements of the environment that are critical for their long-term survival. The challenges brought by climate change are ones that Australia's unique plants and animals must also face if they are to persist and it is through targeted ecophysiological studies that we will best be able to assist them in that process.

Keywords: ecophysiology, desert wallabies, desert kangaroos, Dragon Lizards, Honey Possums, quokka, Barrow Island,Rottnest Island, Shark Bay.

Introductory remarks

I am honoured to have been chosen as a recipient of the Medal of The Royal Society of Western Australia and, since becoming a member in the 1960s, I have long admired the substantial contribution that the Society has made to the study and understanding of our natural environment.

I served on the Council of The Royal Society of Western Australia in the 1970s and was pleased that the Society published a major two-part issue on research on Rottnest Island in 1983 (Bradshaw 1983). In that issue, my former supervisor, Bert Main contributed a paper on macropod ecophysiology, showing that the field of ecophysiology had well and truly arrived (Main 1983). It is amusing to note too that Bert later reports a physiologist telling him that …*"it was quite improper to prostitute physiology as a field study in order to interpret the ecology of species."* (Main 1995)

The discipline of physiology was created by the Frenchman Claude Bernard in the 19th Century but he dealt with animals only as models for the human condition (Bernard 1878). It was only in the 20th century that physiologists began to study animals in their own right, not just as proxies for humans, and the developing discipline of comparative physiology laid the grounds for the emergence of ecophysiology.

Physiology was taught as part of the medical curriculum at The University of Western Australia and any attempt to translate what was known about humans to animals, was actively discouraged, as I found to my cost when I included my own findings on frogs in a report dealing with the clotting times of blood!

Ecophysiology is the study of the relevance of physiological processes in both plants and animals to ecological constraints imposed by their environment. My own research has focused on Australian animals, specifically reptiles, amphibians, birds and marsupials, but lately, plants have become important in working with the nectarivorous Honey possum. I thought I might try in this talk to give you some examples from my own research showing how an ecophysiological approach can have great relevance to the problem of species conservation.

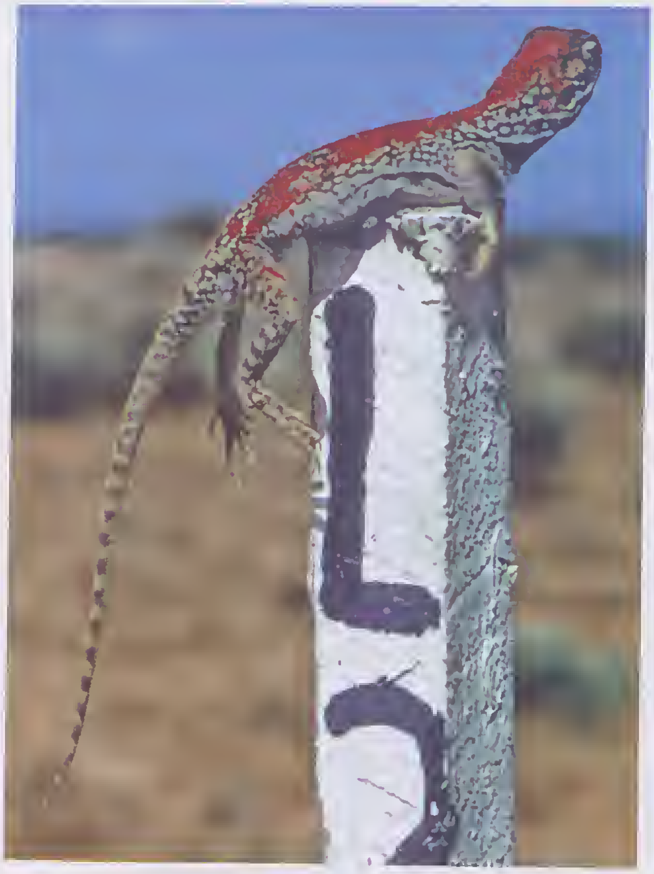


Figure 1. An adult colour-marked male Western netted dragon lizard, *Ctenophorus nuchalis* (formerly *Amphibolurus inermis*) basking on marker post in Shark Bay, some 900 km north of Perth in Western Australia.

Dragon Lizards in Shark Bay

My PhD was on dragon lizards, trying to find out how they contend with the problems of high diurnal temperatures and the lack of water in arid and semi-arid parts of Western Australia (Bradshaw 1970, 1971). I realised when I completed my degree, however, that I could not answer any of the ecological questions that I had posed without a knowledge of the animals' internal control systems – and that meant hormones. I thus spent the next 4 years in overseas laboratories, in Sheffield in the UK, in Paris in France and in Riverside in California, learning and developing new methods to measure steroid hormones from the adrenal glands in extremely small volumes of blood (Bradshaw & Fontaine-Bertrand 1968; Licht & Bradshaw, 1969; Bradshaw & Fontaine-Bertrand 1970). I then applied this in a 10-year study of the Western netted dragon, *Ctenophorus nuchalis*, at Shark Bay (Figure 1). To my amazement I found that these large 40–50g lizards only lived a single year and that they died each summer from a combination of dehydration and starvation to which they seemed ill adapted (Bradshaw 1981). The late Karl Zwicky, one of my colleagues in the Zoology Department, used to say to me ..."why bother studying desert animals because all you find out is that they are adapted to the desert!". Well here was a case where they were not!

By measuring their rates of water turnover with tritiated water and their adrenal physiology at different times of the year I was able to show that they solved the problem of surviving in a fairly hostile environment, not by adapting their adult physiology, but simply by avoiding the problem. They are born in February each year, grow very rapidly and breed in the spring, lay their eggs in the soil and then die. By measuring rates of

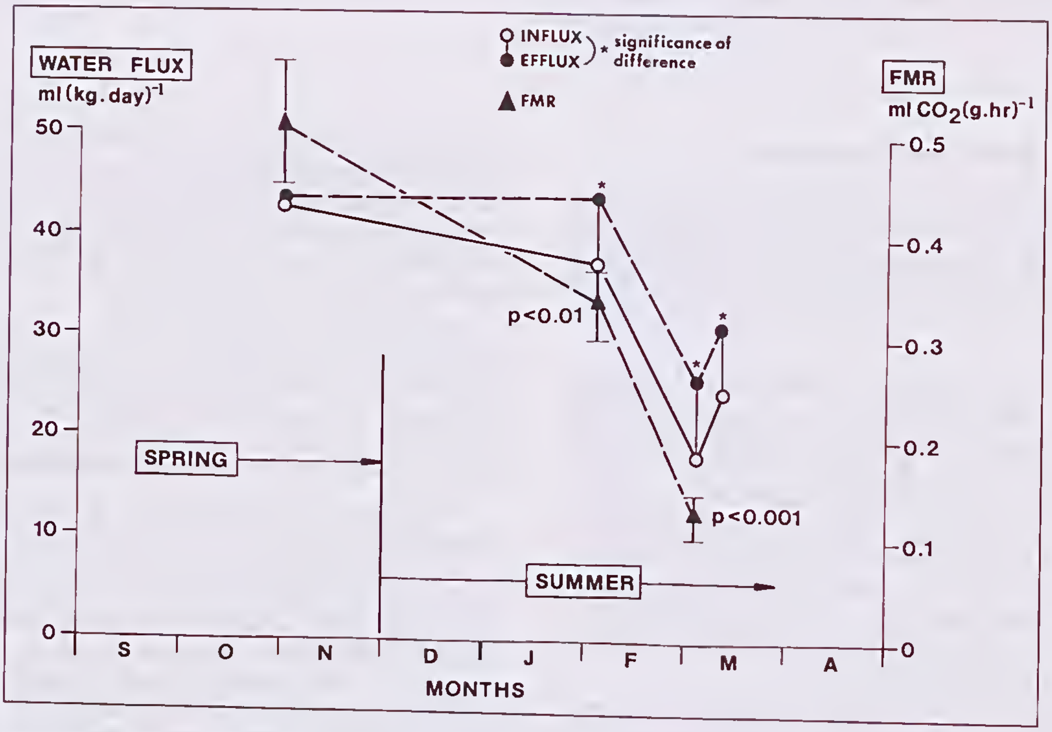


Figure 2. Seasonal changes in rate of turnover of water (influx and efflux) and field metabolic rate (FMR in ml CO$_2$ (g.hr)$^{-1}$) using doubly-labelled water of *Ctenophorus nuchalis* at Shark Bay showing a significant decline in both parameters in late summer (from (Bradshaw 1986).

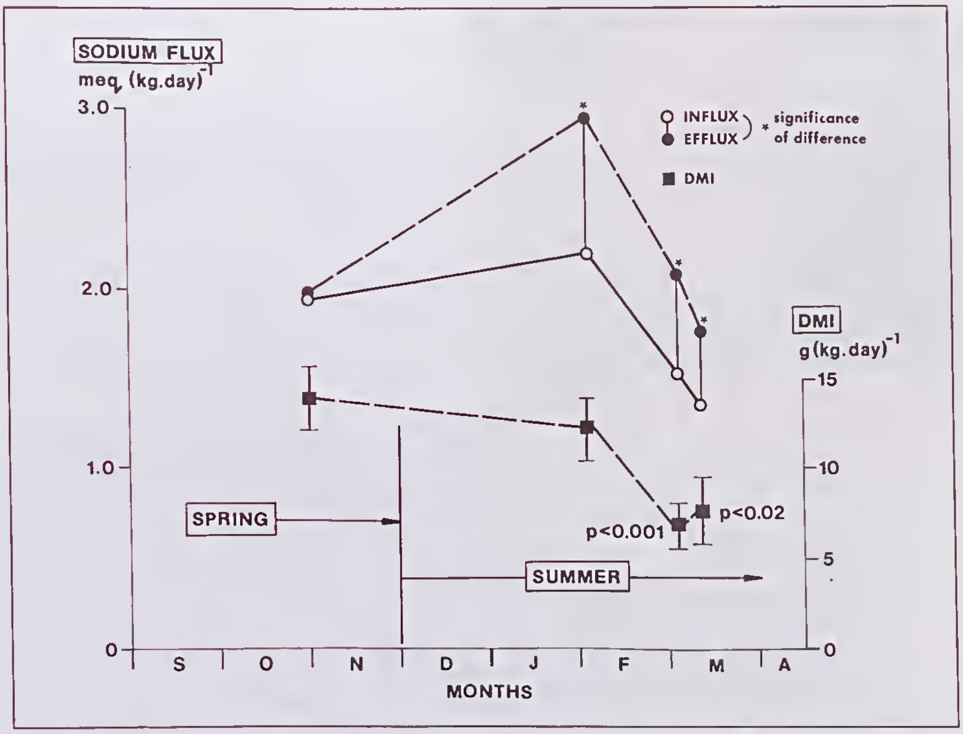


Figure 3. Seasonal variations in rate of sodium turnover (^{22}Na) and estimated dry-matter intake (DMI in g (kg.d)$^{-1}$) of *Ctenophorus nuchalis* at Shark Bay in Western Australia (from (Bradshaw 1986).

turnover of oxygen, water and sodium, I was able to show that their water balance becomes progressively more negative as summer progresses (i.e. efflux > influx) and their field metabolic rate (FMR) declines by more than 80% (Figure 2). Simultaneous measurements of rates of sodium turnover show that the lizards are feeding less and their dry matter intake (DMI) falls to starvation levels by late autumn (Figure 3). The population thus survives in the form of eggs over the hottest part of the year and the young emerge each year into an environment devoid of competing adults! (Bradshaw 1981, 1986; Nagy & Bradshaw 1995)

This was somewhat of an epiphany for me and I went on to look much more critically at the received wisdom – that reptiles represent the acme of vertebrate adaptation to a desert environment (Bradshaw 1987). My conclusion, which I expounded in my first book in 1986, was that reptiles worldwide do not show any specific adaptations to their preferred desert habitat – rather their basic reptilian constitution, with low rates of metabolism, a reliance on external sources of heat and low rates of water turnover, pre-adapt or 'exapt' them in the modern terminology to succeed in desert environments where vital resources are very limited (Bradshaw 1986, 1988).

Quokkas on Rottnest Island

My predecessor, Harry Waring, catapulted the quokka to fame as one of the world's most studied marsupials, with Rottnest Island its home. My interest was stimulated when John Iveson and I discovered that they were riddled with salmonellosis (Iveson & Bradshaw 1973) and that playing with quokkas could have serious health consequences! The story at that time was that quokka numbers had increased dramatically following a devastating fire in 1956 and that the population was now controlled by starvation, with particularly young and old animals, dying over the summer months. In order to test this, I started monitoring the physiological condition of animals on Rottnest every 6 weeks, measuring particularly the level of adrenal hormones in the blood. A negative relationship between plasma steroid levels and body condition was the first indication that quokkas were experiencing stress over summer (Figure 4). We found that they did not die then, however, but at the break of the winter season when low plasma cortisol levels led to hypoglycaemia, with the quokkas unable to

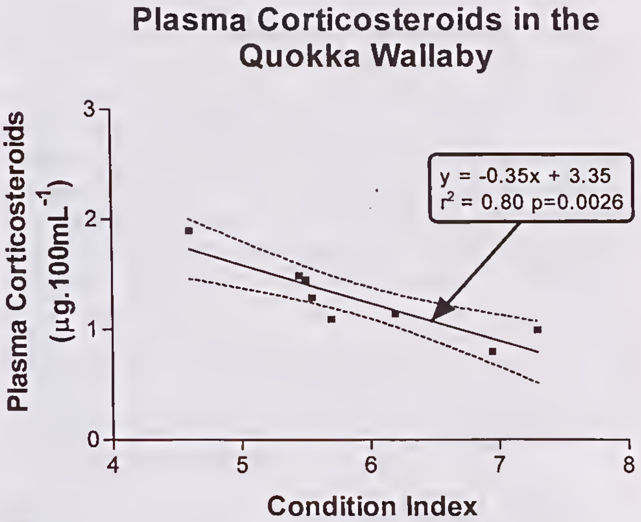


Figure 4. Regression of the concentration of plasma corticosteroid hormones against the body condition index of quokkas on Rottnest Island in late summer. The negative regression is statistically significant, with $r^2 = 0.80$ and $P = 0.0026$ (from Miller & Bradshaw 1979).

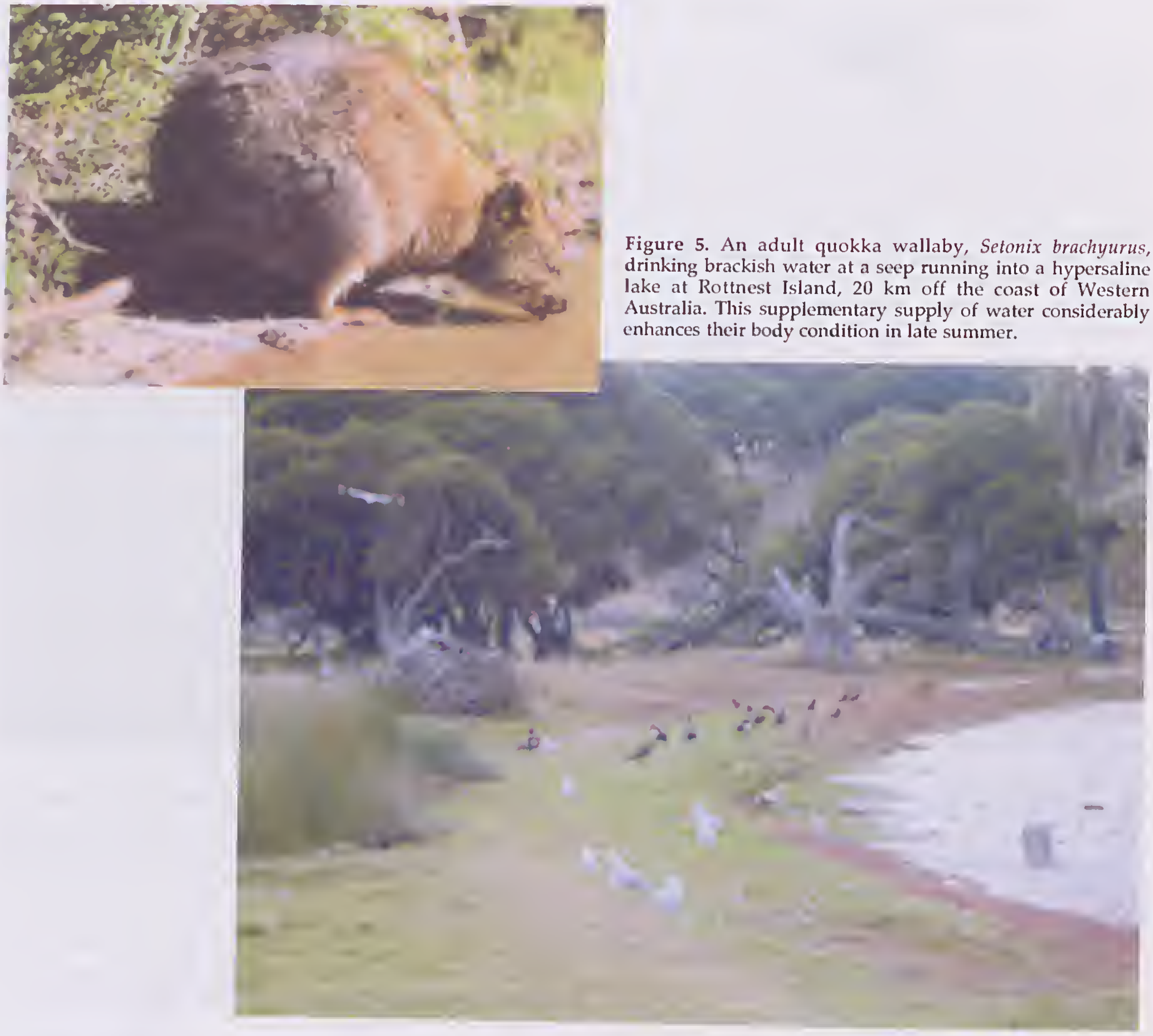


Figure 5. An adult quokka wallaby, *Setonix brachyurus*, drinking brackish water at a seep running into a hypersaline lake at Rottnest Island, 20 km off the coast of Western Australia. This supplementary supply of water considerably enhances their body condition in late summer.

maintain their normal body temperature in the cold weather (Miller & Bradshaw 1979).

A later study with a group of Honours students, in which we measured for the first time the levels of the pituitary antidiuretic hormone (ADH) that controls water loss from the kidney, showed the vital importance of even small quantities of brackish water for the long-term survival of quokkas in the degraded environment that is now Rottnest Island. Quokkas living around the fringes of Lake Baghdad, with access to brackish water from fresh-water seeps flowing into the hypersaline lake (Figure 5), were in substantially better condition than animals from West End, which showed signs of dehydration in late summer (Table 1), despite maximally elevated levels of ADH and reduced rates of urine production (Jones *et al.* 1990).

This study led me to try to clarify the rather confused literature on what is stress and how to measure it. Stress

Table 1

Water and electrolyte balance of quokkas (*Setonix brachyurus*) in late summer on Rottnest Island, Western Australia (from Jones, Bradshaw, Fergusson and Watts 1990)

Parameter	West End	Lake Baghdad	Significance
Body Mass (kg)	2.09 ± 0.15	2.66 ± 0.13	$P<0.01$
Condition Index	5.33 ± 0.26	6.45 ± 0.25	$P<0.01$
Plasma ADH ($pg.mL^{-1}$)	89.2 ± 19.5	35.6 ± 15.8	$P<0.05$
Urine Production ($mL.kg^{-1}.day^{-1}$)	20.35 ± 2.75	58.18 ± 13.79	$P<0.02$
Urine Osmolality ($mOsm.kg^{-1}$)	1253.1 ± 44.7	968.7 ± 101.3	$P<0.02$
Plasma Osmolality ($mOsm.kg^{-1}$)	301.3 ± 6.0	279.8 ± 2.4	$P<0.005$

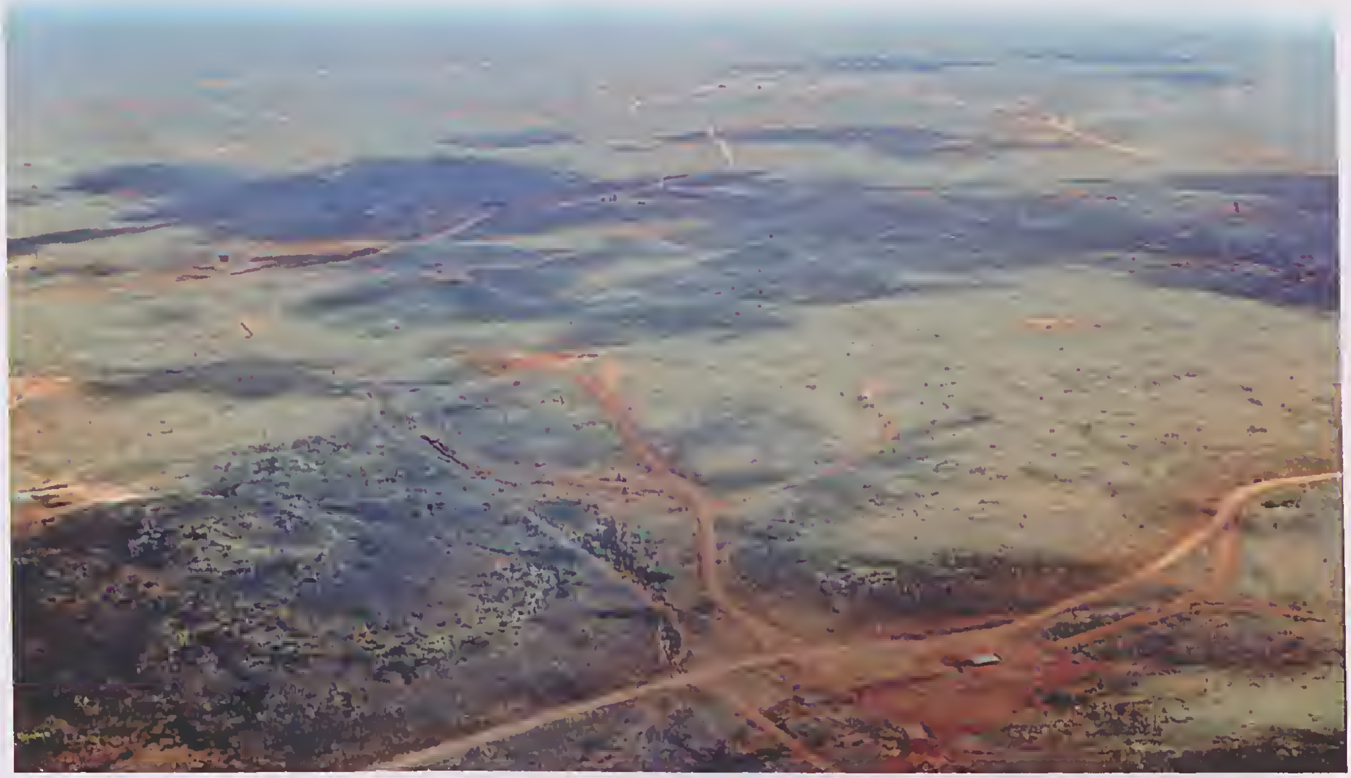


Figure 6. An aerial view of Barrow Island during the dry season of the year. The 30,000 ha island is located some 80 km off the arid Pilbara coast of Western Australia, approximately 1400 km north of Perth.

was originally defined, somewhat poetically, by Hans Selye in 1952 as *"a state of non-specific tension in living matter"* (Selye 1952) and later attempts at a definition have not been much better with *"any factor that inhibits growth and reproduction in a population"* (Brett 1958) or *"an environmental condition that, when first applied, impairs Darwinian fitness"* (Sibley & Calow 1986) – effectively confusing stressors with stress! I tried to simplify the dual problems of identifying stress when it occurs and then measuring its intensity by defining it as *"the physiological consequences of demands that exceed an animal's regulatory capacities"* (Bradshaw 1997; Bradshaw 2003)). This means that one should be able to identify stress by the presence of a significant perturbation of the animal's *milieu intérieur* from whatever is considered to be the 'normal' state, despite the maximum deployment of regulatory systems (Bradshaw 2003). This operational definition turns out to fit in well with the new concepts of 'allostasis' and 'allostatic load' introduced by (McEwen 1998) and (McEwen & Wingfield 2003) and it corresponds to their case of 'allostatic overload' (Bradshaw 2007).

Desert Wallabies and Kangaroos on Barrow Island

Barrow Island off the Pilbara coast is Australia's foremost A Class Nature Reserve with a total of 14 species of mammals, 8 of them marsupial and with 4 of these extinct, or virtually so, on the mainland of Australia (Figure 6). I had a wonderful opportunity to study, with a group of colleagues, the ecophysiology of a whole suite of vertebrates on Barrow Island in the 1990s, courtesy of a grant from the Australian Research Council. By capturing and recapturing the same individuals, we were able to measure, simultaneously, the field metabolic rate (FMR) of a range of vertebrate animals with oxygen-18 and thus compare their metabolic footprints, using oxygen consumption as a common currency (Bradshaw 1992). Using this ecophysiological approach it was clear that the Golden bandicoot, *Isoodon auratus*, was by far the most important animal metabolically on the island (Bradshaw *et al.* 1994).

Attempts to recapture rock wallabies (*Petrogale lateralis*) on Barrow, which live in caves of the west coast of the island, were not overly successful and, with Keith Morris' help, we initiated a study of the related species, *Petrogale rothschildi*, on Enderby Island in the Dampier Archipelago (Figure 7). Our aim was to compare their ecophysiology with Spectacled hare wallabies (*Lagorchestes conspicillatus*), which are very common on Barrow, and spend the daylight hours in spinifex tussocks where they are exposed to high environmental temperatures (Figure 8).

Our results were quite intriguing. Both species reduced dramatically their rates of water turnover and urine production in summer and both also showed no sign of any perturbation of their *milieu intérieur* or 'internal environment', even under the most trying conditions. The extent to which the Hare wallabies are

Figure 7. Rothschild's rock wallaby, *Petrogale Rothschildi,* disappearing over a hill on Enderby Island in the Dampier Archipelago following a physiological interrogation.

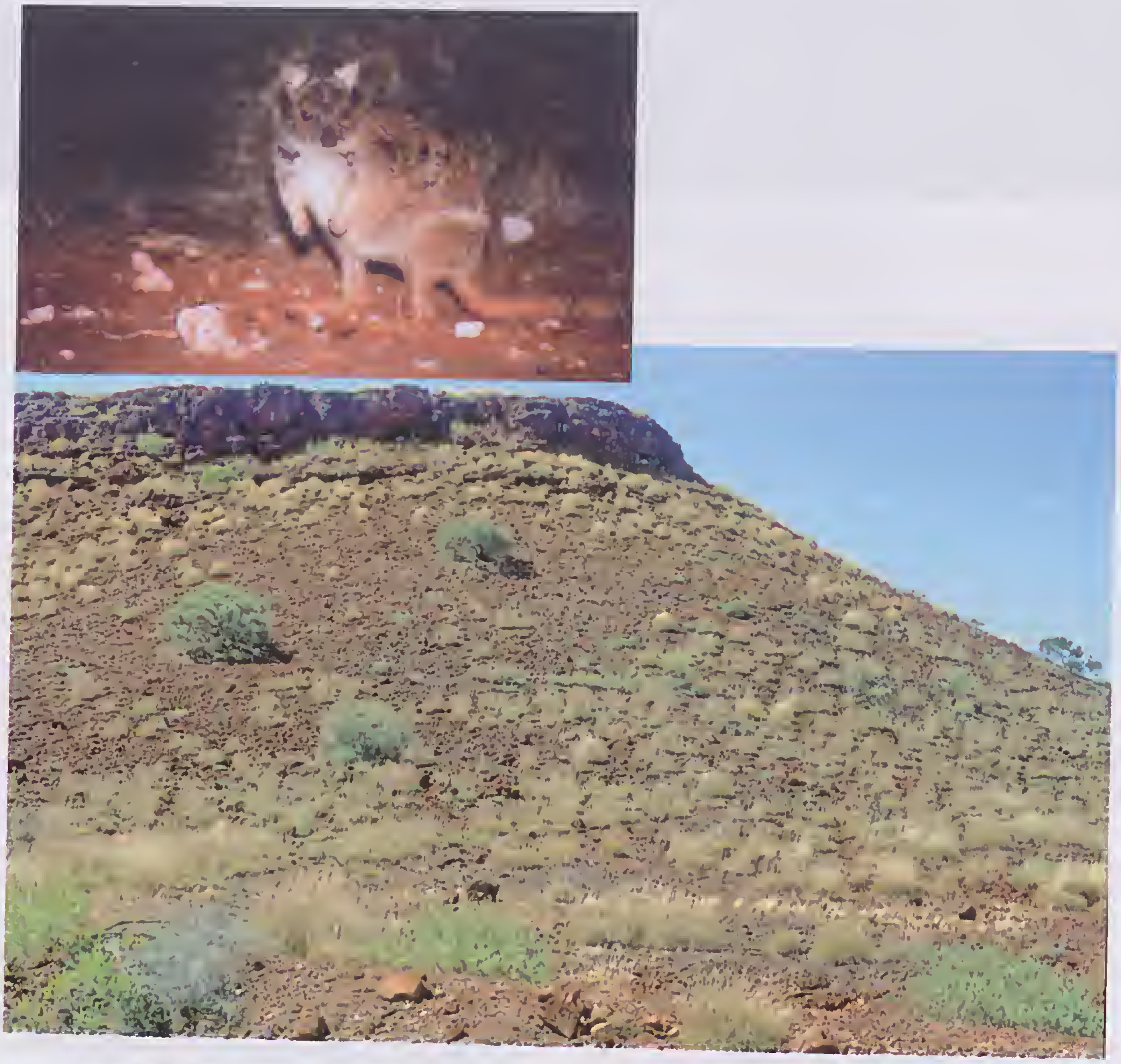


Figure 8. The Spectacled hare wallaby, *Lagorchestes conspicillatus,* on Barrow Island and a section of habitat showing the clumps of spinifex (*Triodia spp.*) in which the wallabies shelter during the heat of the day

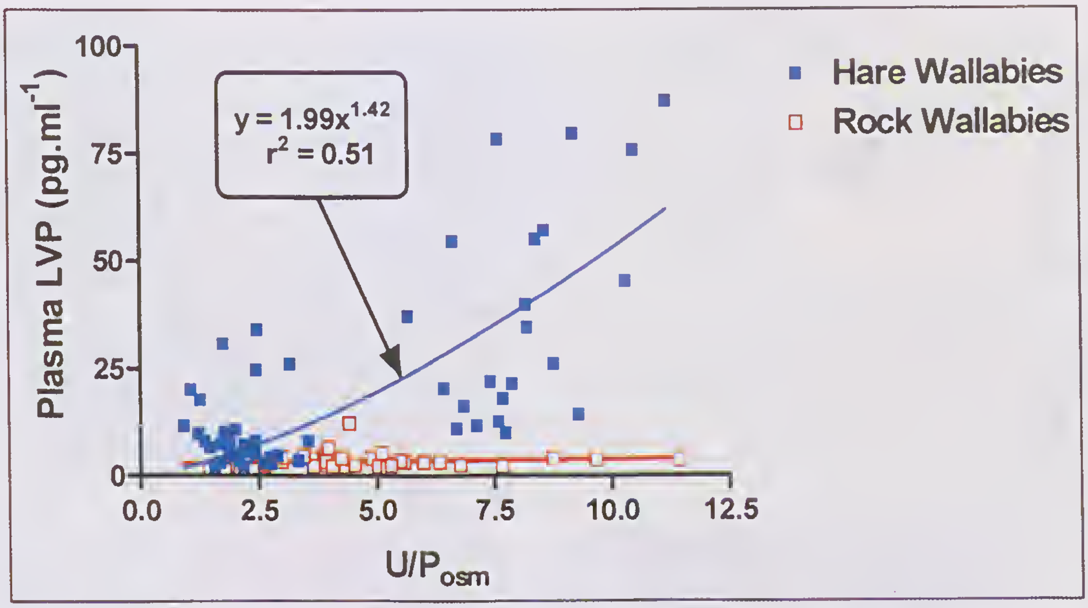


Figure 9. Variation in circulating levels of lysine vasopressin (LVP) in Hare wallabies and Rock wallabies as function of the concentrating activity of the kidney (U/P_{osm}) (from Bradshaw, Morris & Bradshaw 2001).

able to conserve water is quite remarkable and their rate of water turnover is the lowest of any desert mammal measured worldwide (Bakker & Bradshaw 1989; Nagy & Bradshaw 2000; King & Bradshaw 2008). It was when we looked in detail at the means by which the two species were conserving water loss from their kidneys that an amazing difference emerged (Figure 9). The Hare wallabies displayed a typical mammalian response to lack of water. Plasma levels of ADH increased (as in quokkas on the West End of Rottnest), urine volume declined and its osmotic pressure increased as water was abstracted by the renal concentrating mechanism (Table 2). When we looked at the Rock wallaby on the other hand, we found that ADH levels did not change from wet to dry season and, although urine volumes declined in the dry season, the urine did not get more concentrated (Table 3). What the Rock wallabies were doing was responding somewhat like reptiles and simply reducing their rate of urine production by cutting down blood flow to the kidneys. Less urine was thus being produced, and less water being lost, but without any change in its concentration. We thus appear to have discovered the first instance of a mammal completely lacking an hormonally-controlled regulation of kidney function as shown in Figure 9, (Bradshaw *et al.* 2001). This discovery serves to underline the vital importance of the cool and humid cave environment in which the Rock wallabies shelter during the heat of the day. If this were to be destroyed they lack the physiological adaptations that would enable them to survive in this arid environment (King & Bradshaw 2008).

Table 2

A summary of kidney function in the Spectacled hare wallaby, *Lagorchestes conspicillatus*, on Barrow Island in wet and dry seasons (from Bradshaw, Morris and Bradshaw, 2001)

Season	C_{PAH}	C_{IN}	C_{OSM}	C_{H2O}	V	FR_{H2O}	U/P_{OSM}	ECFV (%)	LVP
DRY	6.0 ± 0.4	1.5 ± 0.1	57.5 ± 3.2	-50.5 ± 2.8	6.9 ± 2.2	99.6 ± 0.1	8.4 ± 0.3	18.3 ± 0.7	32.7 ± 5.6
WET	9.0 ± 1.3	2.7 ± 0.2	91.6 ± 6.9	-39.6 ± 3.9	50.2 ± 4.5	97.8 ± 0.3	2.0 ± 0.1	18.2 ± 1.4	10.7 ± 1.8
P =	0.04	0.001	0.001	0.059	.001	0.001	0.001	NS	0.001

Table 3

A summary of kidney function in Rothschild's rock wallaby, *Petrogale rothschidi*, on Enderby Island in wet and dry seasons (from Bradshaw, Morris and Bradshaw, 2001)

Season	C_{PAH}	C_{IN}	C_{OSM}	C_{H2O}	V	FR_{H2O}	U/P_{OSM}	ECFV (%)	LVP
DRY	5.8 ± 0.1	0.8 ± 0.1	34.8 ± 6.7	-28.1 ± 5.4	6.7 ± 1.4	99.4 ± 0.1	5.4 ± 0.7	12.2 ± 0.5	3.5 ± 1.5
WET	7.9 ± 0.5	1.3 ± 0.1	59.2 ± 6.2	-46.3 ± 4.5	14.3 ± 2.1	99.2 ± 0.1	5.4 ± 1.1	18.8 ± 1.1	6.0 ± 0.9
P =	.008	0.005	0.02	0.02	0.01	NS	NS	0.001	NS

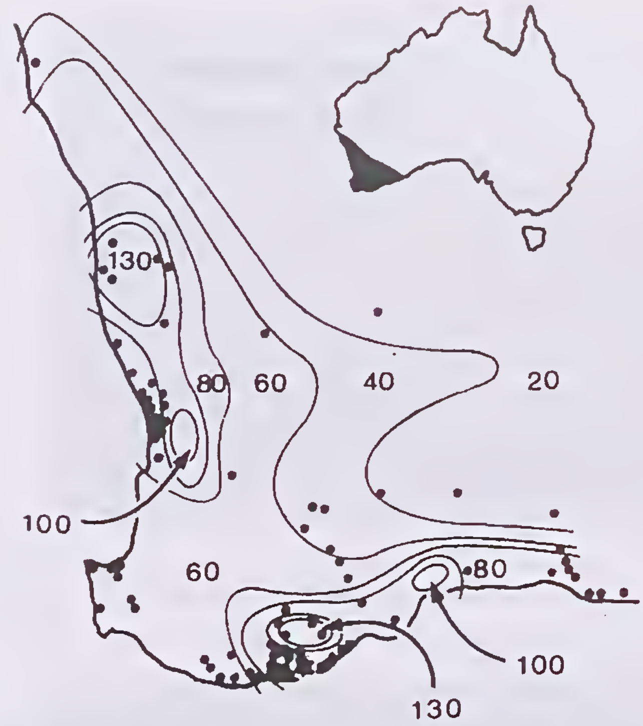


Figure 10. The past and present distribution of the Honey possum, *Tarsipes rostratus,* in south-western Western Australia (from Wooller, Russelll and Renfree, 1984). Each dot indicates a specimen in the Western Australian Museum collection and the lines are isoflors joining points of equal plant species richness within the family Proteaceae (after Hopper 1979).

Although the Barrow Island fauna displays an impressive level of physiological adaptation to the exigencies of its arid environment, our study did reveal one instance of probable stress in the dwarf euro kangaroo, *Macropus robustus isabellinus.* Evidence was found of a significant change in the animal's *milieu intérieur* on one occasion on Barrow Island: in November 1994, following a protracted 8-month drought when a total of only 54 mm of rain was recorded on the island. Euros had significantly elevated levels of plasma osmolality, cortisol, ADH, and a reduced eosinophil count (King & Bradshaw 2010). This suggests that these animals may have been dehydrated, despite the operation of appropriate physiological responses to water deprivation. Lower eosinophil counts also suggest that immune function may have been suppressed as a result of the elevated corticosteroid levels. Comparisons with the mainland sub-species of the euro also revealed the presence of a non-generative normocytic hypochromic anaemia in Barrow Island euros that potentially compromises their aerobic capacity (Billiards *et al.* 1999; Ogawa *et al.* 2000) and adds to their vulnerability.

Honey Possums

I should like to close with a brief mention of our current research project on the tiny nectarivorous Honey possum, *Tarsipes rostratus,* in Scott National Park in the extreme southwest of Western Australia (Figure 10). The southwest corner of Western Australia contains

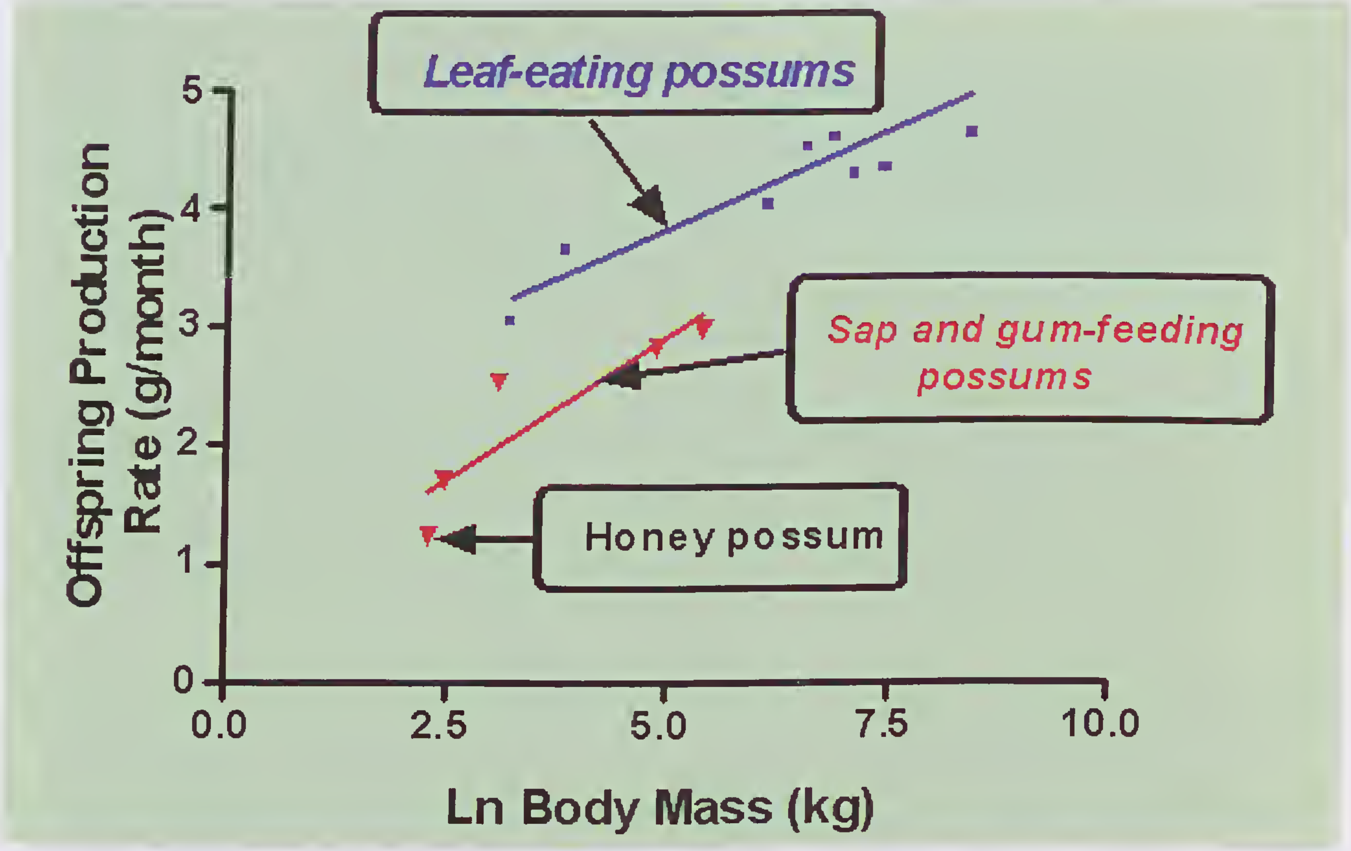


Figure 11. Relationship between the offspring production rate, in grammes per month, and female adult body mass in leaf-eating *versus* sap, nectar and gum-feeding possums (from Smith & Lee 1984).

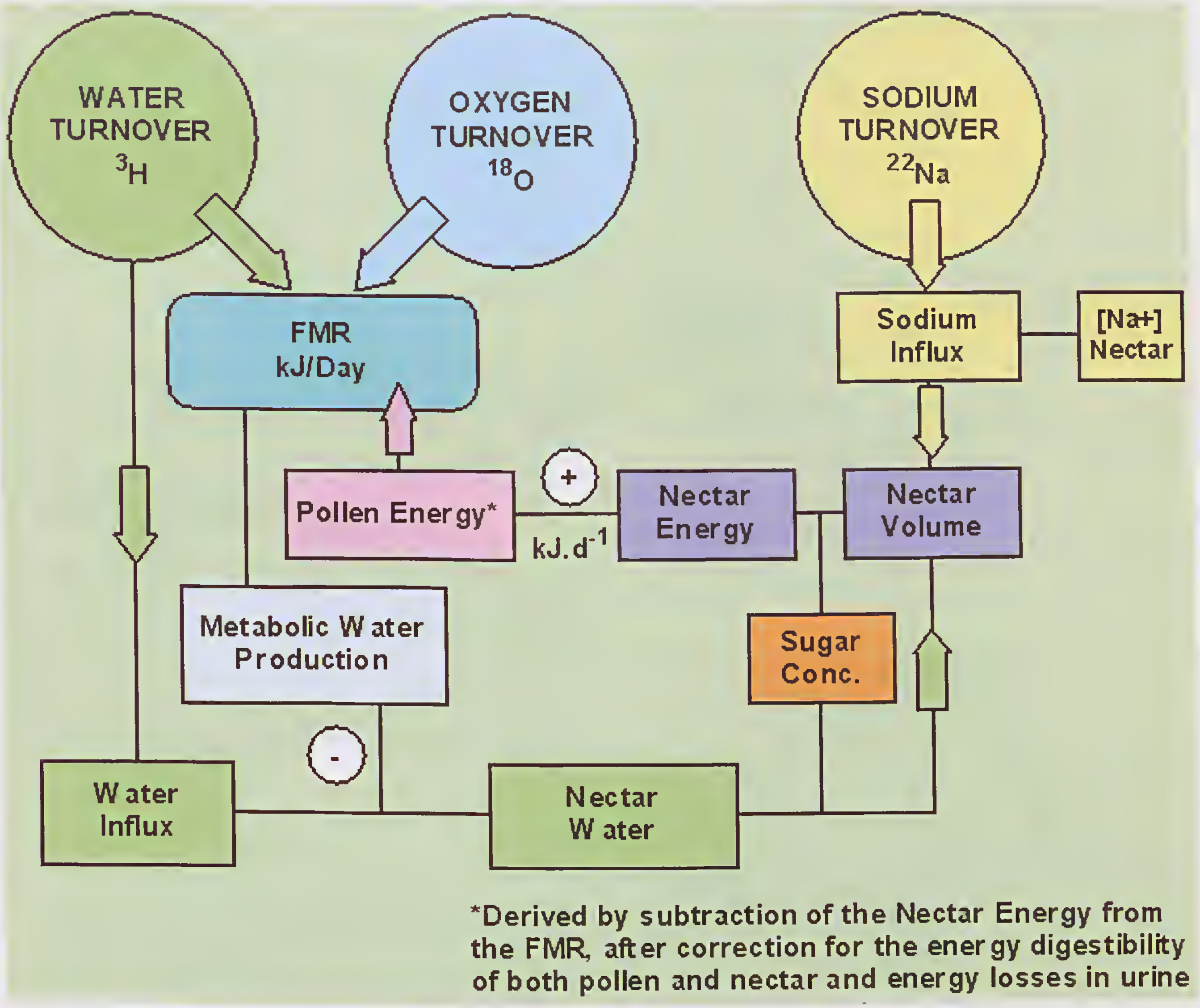
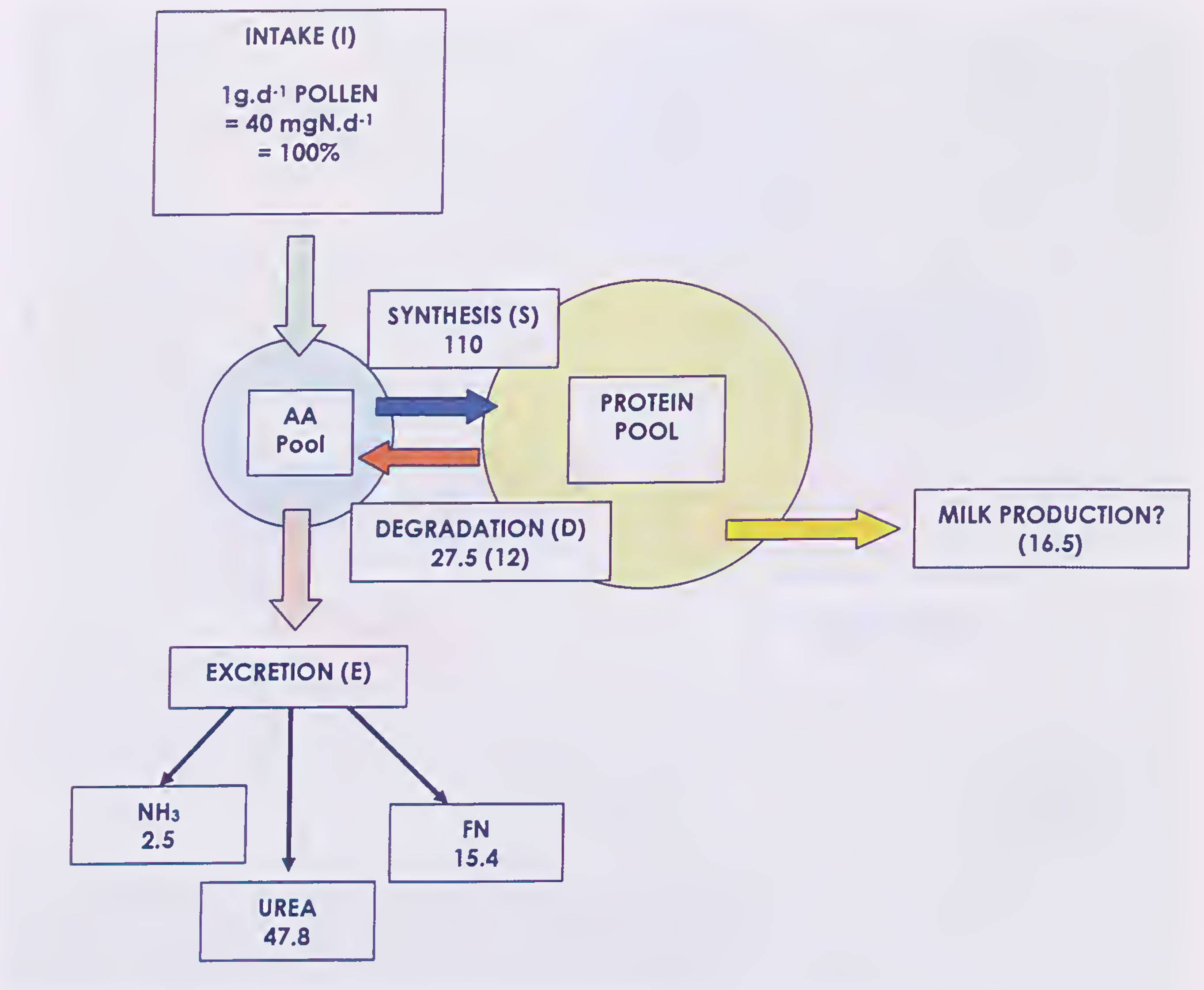

WATER TURNOVER ^3H
OXYGEN TURNOVER ^{18}O
SODIUM TURNOVER ^{22}Na
FMR kJ/Day
Sodium Influx
[Na+] Nectar
Pollen Energy*
+
kJ.d^{-1}
Nectar Energy
Nectar Volume
Metabolic Water Production
Sugar Conc.
-
Water Influx
Nectar Water
*Derived by subtraction of the Nectar Energy from the FMR, after correction for the energy digestibility of both pollen and nectar and energy losses in urine

Figure 12. Schema of the rationale used to measure rates of pollen and nectar intake using stable and radioactive isotopes in free-ranging Honey possums (*Tarspies rostratus*) in Scott National Park. Nectar intake is estimated independently from influxes of sodium-22 and tritium, the latter corrected for metabolic water production (MWP) and averaged. Pollen intake is estimated by subtraction of the nectar energy intake from the field metabolic rate (FMR) (from Bradshaw & Bradshaw 1999).

Australia's only biodiversity hotspot threatened by habitat loss and degradation (Myers *et al.* 2000; Mittermeier *et al.* 2004) and the Honey possum is extremely vulnerable, depending as it does on kwongan vegetation now threatened by fire and spreading *Phytophthora cinnamomi* infections.

We set out to test an old hypothesis in the literature that the relatively low reproductive rate of Honey possums, compared with leaf-eating species, may be due to their high carbohydrate diet (nectar) that could be deficient in protein (Figure 11) (Smith & Lee 1984). We needed to know how much nectar and pollen Honey possums actually consume daily in their natural habitat and, after much work at Lucas Heights, we developed an isotopic method to determine this (Figure 12) (Bradshaw *et al.* 1987; Bradshaw & Bradshaw 1999). We found that a 9 g possum consumes, on average, 7 ml of nectar and 1 g of pollen – virtually its own body mass daily. Although pollen had been thought in the literature to be indigestible, it is a rich source of nitrogen and 1 g of pollen per day provides approximately 10 times the Honey possum's minimum requirement for nitrogen balance (Bradshaw &Bradshaw 2001). We then went on to measure rates of protein metabolism using ^{15}N-glycine in lactating females and were able to establish that they divert the very small amounts of nitrogen needed to sustain the developing young *via* the milk by decreasing rates of protein degradation, rather than increasing rates of synthesis when lactating (Bradshaw &Bradshaw 2009) (Figure 13). The Honey possum joey is the size of a grain of rice when born and only weighs 4 mg – only minute amounts of nitrogen are thus needed to be diverted from the mother's diet through the milk to sustain its development. We can calculate that the total amount of nitrogen needed to produce a young of 2.5 g at weaning is only 128 mg, and contrast this with the mother's daily intake of nitrogen on its pollen diet of approximately 40 mg (Bradshaw & Bradshaw 1999).



URINARY N (50.3) + FAECAL N (15.4) = TOTAL N EXCRETION = 65.7

N BALANCE = 34.3

Figure 13. A model of nitrogen flow in relation to dietary intake of 1 g of pollen of 4% nitrogen content for the Honey possum *Tarsipes rostratus*. Dietary intake is set at 40 mg per day and taken as 100% with all other parameters shown as percentage values of this. I = nitrogen intake, E = nitrogen excretion, S = rate of nitrogen synthesis to the protein pool and D = rate of nitrogen degradation from the protein pool. FN = faecal nitrogen excretion, urea = urinary urea excretion and NH_3 = urinary ammonia excretion. The value for D in parentheses (12) represents the lower whole body degradation rate measured in females carrying pouch young (from Bradshaw & Bradshaw 2009).

Although one can never prove hypotheses in science, we have shown that the one initiating this study is false – i.e. that their high carbohydrate diet is deficient in protein and that this accounts for the Honey possum's relatively low reproductive rate when compared with leaf-eating possums (Smith & Lee 1984). It also highlights the fact that this was the wrong question to ask in the first place. Because an animal has a lower rate of reproduction than another does not necessarily mean that its diet is wanting. Rates of reproduction of different species are attuned to their particular environment as a result of natural selection and, unless the population is in obvious decline, the reproductive rate will be adequate to balance mortality and emigration (Stearns 1992). The Honey possum is unusual amongst marsupials in retaining its young with the mother for a longer period of time than similar-sized phalangeroids (Russell 1982; Wooller *et al.* 1984), suggesting that this enhanced parental care involves a longer learning experience that increases their probability of survival and thus compensates for a lower offspring production rate than other small possums.

We now have a quantitative measure of the amounts of energy needed to sustain Honey possums in their kwongan habitat, and the flower resources needed to provide these daily amounts of carbohydrate and nitrogen. The Honey possum is not yet listed as vulnerable or threatened, but it will be soon if the current loss and destruction of its habitat continues unabated. Fire is also one of its main enemies and our long-term study in Scott National Park has shown that recovery after fire is very slow with an estimate of 15–20 years to recover pre-fire population densities (Bradshaw *et al.* 2007).

In summary

I hope that these few examples will give an idea of the power of the ecophysiological approach – bridging two established disciplines to better understand the vital nexus between an animal and its environment – and opening vistas for effective conservation of our unique biodiversity legacy in Western Australia.

Acknowledgements: I have been most fortunate in that my research over the years on sometimes obscure native animals has been generously funded by the Australian Research Council (ARC), and its forerunners ARGC and ARGS. An important group of colleagues, both in Australia and overseas, have greatly assisted me in my researches, as have the many graduate students that I have had the privilege to supervise over a period of forty years at The University of Western Australia. My greatest thanks though go to my most valuable colleague and collaborator, my wife Felicity, and I express my love and thanks to her for her unwavering support in everything I do.

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